

Technical/Agency Draft Revised

Recovery Plan for the

Red-cockaded Woodpecker

(*Picoides borealis*)

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U.S. Fish and Wildlife Service
Southeast Region
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DISCLAIMER

This Recovery Plan Revision outlines the actions that, to the best of current understanding, are necessary to recover red-cockaded woodpeckers. It does not represent the view or official position of any individuals or agencies involved in the development of the plan, other than the U.S. Fish and Wildlife Service. It represents official policy of the U.S. Fish and Wildlife Service only after it has been signed by the regional director as approved. This revision is subject to further modification as dictated by new findings, changes in species status, and completion of recovery tasks. Implementation of this plan is the responsibility of federal and state management agencies in the areas where the species occurs. Implementation is done through incorporation of management guidelines identified within this Recovery Plan Revision into agency decision documents. Decision documents, as defined by the National Environmental Policy Act (NEPA), are subject to the NEPA process for public review and alternatives selection.

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STANDARD ABBREVIATIONS

The following standard abbreviations for units of measurement are found throughout this document:

cm = centimeters	in = inches	sq. m = square meters
m = meters	ft = feet	sq. ft = square feet
km = kilometers	mi = miles	dbh = diameter at breast height
ha = hectares	ac = acres	
g = grams	oz = ounces	

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The process of revising the 1985 red-cockaded woodpecker recovery plan began in August 1995, when potential recovery team members were identified. In January 1996, 15 potential members were asked to participate on the team; all accepted. The first team meeting was held in March 1996; two additional meetings, each one week long, were held in April and December, 1997. Between March 1996 and March 2000, team members individually spent hundreds of hours working on the revision, including participation in team meetings, writing, and reviewing. The U.S. Fish and Wildlife Service is very appreciative of the time and hard work put forth by team members during this process. Combined, the team has approximately 257 years of red-cockaded woodpecker experience in the private, state, and federal sectors. Their professional experiences with red-cockaded woodpeckers have included research, population and habitat management, and regulatory and policy responsibilities. They have unselfishly contributed their knowledge, time, and expertise to the many challenges of the recovery plan revision process. The U.S. Fish and Wildlife Service thanks each of them for their contributions and is grateful to have worked with such a team.

By April 1999, much of the Introduction had been drafted. However, several major tasks remained to be accomplished. These tasks included writing the Recovery section and the Executive Summary, compiling and editing the chapters of 16 different authors, and creating tables, Literature Cited, and Index. In April 1999, the U.S. Fish and Wildlife Service hired Ms. Susan Daniels as a wildlife biologist and recovery team member to take the lead on completing these tasks. The recovery team and U.S. Fish and Wildlife Service are proud to extend special thanks to “one of our own” for her continuous hard work during the past year on this challenging project. Sue has done an outstanding job of assembling and completing the revision.

Ultimately, recovery of the red-cockaded woodpecker is and will be dependent upon the on-the-ground hard work of biologists, foresters, technicians, researchers, and land managers working on the private, state, and federal properties where the birds survive. During the past five years, many of these individuals have been asked to supply information including population and habitat data, maps, and management costs. The U.S. Fish and Wildlife Service and the recovery team are particularly grateful to these many individuals for their timely and reliable responses to our requests. They have supplied a tremendous amount of information for this document.

TABLE OF CONTENTS

ACKNOWLEDGMENTS.....	iii
EXECUTIVE SUMMARY.....	vi
PART I. INTRODUCTION.....	1
1. LISTING.....	1
A. REASONS FOR LISTING.....	1
B. CURRENT THREATS.....	4
2. GENERAL BIOLOGY AND ECOLOGY.....	9
A. TAXONOMY AND SPECIES DESCRIPTION.....	9
B. SOCIOBIOLOGY AND COOPERATIVE BREEDING.....	10
C. POPULATION AND SPECIES VIABILITY.....	21
D. CAVITY TREE AND CLUSTER ECOLOGY.....	31
E. FORAGING ECOLOGY.....	40
F. COMMUNITY ECOLOGY:	
CAVITY KLEPTOPARASITISM, CAVITY ENLARGEMENT, AND PREDATION.....	55
G. THE ROLE OF FIRE IN SOUTHERN PINE ECOSYSTEMS.....	60
3. MANAGEMENT TECHNIQUES.....	64
A. POPULATION MONITORING.....	64
B. CAVITY MANAGEMENT: ARTIFICIAL CAVITIES AND RESTRICTOR PLATES.....	73
C. PREDATOR AND CAVITY KLEPTOPARASITE CONTROL.....	82
D. TRANSLOCATION.....	84
E. SILVICULTURE.....	88
F. PRESCRIBED BURNING.....	94
G. HABITAT RESTORATION.....	98
H. ECOSYSTEM MANAGEMENT.....	103
4. CURRENT STATUS AND CONSERVATION INITIATIVES.....	106
A. PRIVATE LANDS.....	106
B. STATE LANDS.....	115
C. FEDERAL LANDS.....	120
PART II. RECOVERY.....	126
5. RECOVERY GOAL.....	126
6. RECOVERY CRITERIA.....	126
A. DELISTING.....	126
B. DOWNLISTING.....	128
7. RECOVERY UNITS.....	129
8. MANAGEMENT GUIDELINES.....	143
A. RECOMMENDED RATE OF INCREASE AND USE OF RECRUITMENT CLUSTERS.....	143
B. POPULATION MONITORING.....	144
C. CAVITY MANAGEMENT, ARTIFICIAL CAVITIES, AND RESTRICTOR PLATES.....	145
D. CLUSTERS AND CAVITY TREES.....	147
E. PREDATORS AND CAVITY KLEPTOPARASITES.....	149
F. TRANSLOCATION.....	150
G. FORAGING HABITAT.....	154

H. SILVICULTURE.....	161
I. PRESCRIBED BURNING.....	162
9. RECOVERY TASKS	166
10. IMPLEMENTATION SCHEDULE AND ESTIMATED COSTS.....	171
 LITERATURE CITED.....	 188
GLOSSARY OF TERMS	216
INDEX	222
APPENDIX 1. PERMITS, TRAINING, AND COMPLIANCE REQUIREMENTS	226

LIST OF TABLES

TABLE 1. Comparison of techniques for monitoring red-cockaded woodpecker populations.	67
TABLE 2. Estimated number of active red-cockaded woodpecker clusters in 1998 on lands owned by state governments.	116
TABLE 3. Status of red-cockaded woodpecker populations on public lands, by state ...	117
TABLE 4. Role in recovery, current size, and estimated population trend for red-cockaded woodpecker populations on federal lands, listed by responsible agency.	124
TABLE 5. Populations of red-cockaded woodpeckers on publicly owned lands and their designated role in recovery, by recovery unit.	138
TABLE 6. Rationale for foraging guidelines based on habitat structure (recovery standard).	158
TABLE 7. Implementation schedule and estimated costs by recovery task.....	173
TABLE 8. Estimated time for each recovery population to attain size required for delisting the species.	177
TABLE 9. Estimated time for each recovery population to attain size required for downlisting the species	179
TABLE 10. Estimated annual cost and schedule for implementation of recovery task 1.1.2 (<i>maintain four suitable cavities in each active cluster</i>)	180
TABLE 11. Estimated annual cost and schedule for implementation of recovery task 1.2.3 (<i>provision recruitment clusters equal to 10 percent of population, 4 artificial cavities each</i>).....	184
TABLE 12. Estimated annual cost for implementation of recovery task 1.7 (<i>burn entire area managed for red-cockaded woodpeckers at least every 3 to 5 years</i>).....	186

LIST OF FIGURES

FIGURE 1. Diagram of Copeyon-drilled cavity	75
FIGURE 2. Diagram of Copeyon-drilled start.	75

EXECUTIVE SUMMARY

CURRENT STATUS

The red-cockaded woodpecker (*Picoides borealis*) is a federally listed endangered species endemic to open, mature and old-growth pine ecosystems in the southeastern United States. Currently, there are an estimated 12,500 red-cockaded woodpeckers living in roughly 5,000 family groups across twelve states. This is less than 3 percent of estimated abundance at the time of European settlement. Red-cockaded woodpeckers were given protection with the passage of the Endangered Species Act in 1973. Despite this protection, all monitored populations (with one exception) declined in size throughout the 1970's and into the 1980's. In the 1990's, in response to intensive management based on a new understanding of population dynamics and new management tools, most populations were stabilized and many showed increases. Other populations remain in decline, and most have small population sizes. Our major challenge now is to bring about the widespread increases in population sizes necessary for recovery.

BASIC ECOLOGY AND POPULATION DYNAMICS

Red-cockaded woodpeckers are a cooperatively breeding species, living in family groups that typically consist of a breeding pair with or without one or two male helpers. Females may become helpers, but do so at a much lower rate than males. The ecological basis of cooperative breeding in this species is unusually high variation in habitat quality, due to the presence or absence of a critical resource. This critical resource is the cavities that red-cockaded woodpeckers excavate in live pines, a task that commonly takes several years to complete.

Red-cockaded woodpeckers exploit the ability of live pines to produce large amounts of resin, by causing the cavity tree to exude resin through wounds, known as resin wells, that the birds keep open. This resin creates an effective barrier against climbing snakes. Longleaf pine is a preferred tree species for cavity excavation because it produces more resin, and for a longer period of time, than other southern pines.

Group living has profound influence over population dynamics. In non-cooperatively breeding birds, breeders that die are replaced primarily by the young of the previous year. Thus, variation in reproduction and mortality can have strong, immediate impacts on the size of the breeding population. However, in red-cockaded woodpeckers and other cooperative breeders, a large pool of helpers is available to replace breeders. As a result, the size of the breeding population is not strongly affected by how many young are produced each year, or even on how many breeders may die. Because of this, we use the number of potential breeding groups (defined below) rather than number of individuals as our measure of population size.

Because of the cooperative breeding system, red-cockaded woodpecker populations are unusually resistant to environmental and demographic variation, but highly sensitive to the spatial arrangement of habitat. The buffering effect of helpers against annual variation operates only when helpers can readily occupy breeding vacancies as they arise. Helpers do not disperse very far and typically occupy vacancies on their natal territory or a neighboring one. If groups are isolated in space, dispersal of helpers to neighboring territories is disrupted and the buffering effect of the helper class is lost. When this happens, populations become much less likely to persist through time. Also, the cooperative breeding system does not allow rapid natural growth of populations. Colonization of unoccupied habitat is an exceedingly slow process under natural conditions, because cavities take long periods of time to excavate and birds do not occupy habitat without cavities. As forests age and old pines become abundant, rates of natural cavity excavation may increase.

Understanding these three components of the population dynamics of red-cockaded woodpeckers provides us the foundation for recovery efforts: (1) population size and trend are determined by the number of potential breeding groups rather than annual variation in reproduction and survival; (2) the buffering capacity of the helper class must be maintained, by maintaining close aggregations of territories; and (3) colonization of unoccupied habitat will be very slow without management assistance.

HABITAT REQUIREMENTS AND LIMITING FACTORS

Red-cockaded woodpeckers require open pine woodlands and savannahs with large old pines for nesting and roosting habitat (clusters). Large old pines are required as cavity trees because the cavities are excavated completely within inactive heartwood, so that the cavity interior remains free from resin that can entrap the birds. Also, old pines are preferred as cavity trees, because of the higher incidence of the heartwood decay that greatly facilitates cavity excavation. Cavity trees must be in open stands with little or no hardwood midstory and few or no overstory hardwoods. Hardwood encroachment resulting from fire suppression is a well-known cause of cluster abandonment. Red-cockaded woodpeckers also require abundant foraging habitat. Suitable foraging habitat consists of mature pines with an open canopy, low densities of small pines, little or no hardwood or pine midstory, few or no overstory hardwoods, and abundant native bunchgrass and forb groundcovers.

Limiting factors are those that directly affect the number of potential breeding groups, because this is the primary determinant of population size and trend. Several factors currently impact the persistence of breeding groups. Foremost among these are the factors that limit suitable nesting habitat, namely fire suppression and lack of cavity trees. Fire suppression has resulted in loss of potential breeding groups throughout the range of red-cockaded woodpeckers, because the birds cannot tolerate the hardwood encroachment that results from lack of fire. This limitation is addressed through the use of prescribed burning. Lack of cavity trees, and potential cavity trees, limits the number of breeding groups in most populations. This limitation is addressed in the short-term

through cavity management tools such as artificial cavities and restrictor plates, and over the long-term by growing large old trees in abundance.

Another factor directly limiting the number of potential breeding groups is habitat fragmentation and consequent isolation of groups, which results in disrupted dispersal of helpers and failure to replace breeders. This limitation is best addressed through the appropriate placement of clusters of artificial cavities, and implementation of silvicultural practices that minimize fragmentation.

There are several other threats to the existence and recovery of the species, not limiting most populations currently, but which will become more important as the current limitations are addressed. Chief among these are (1) degradation of foraging habitat through fire suppression and loss of mature trees, and (2) loss of valuable genetic resources because of small size and isolation of populations. As currently limiting factors such as lack of cavities are relieved, the continued growth and natural stability of red-cockaded woodpecker populations will depend on provision of abundant, good quality foraging habitat and careful conservation of genetic resources.

POPULATION AND SPECIES VIABILITY

Four types of threats to species and population viability have been identified: genetic stochasticity (consisting of both inbreeding and genetic drift), demographic stochasticity, environmental stochasticity, and catastrophes. We now have some knowledge of population sizes of red-cockaded woodpeckers necessary to withstand these extinction threats, primarily from research performed with a spatially explicit, individually based simulation model of population dynamics developed specifically for this species.

Red-cockaded woodpeckers exhibit inbreeding depression and inbreeding avoidance behaviors. Inbreeding is expected to affect population viability in populations of less than 40 potential breeding groups, and may be a significant factor affecting viability in isolated populations of 40 to 100 potential breeding groups as well. Immigration rates of 2 or more migrants per year can effectively reduce inbreeding in populations of any size, including very small ones.

Effects of demographic stochasticity on population viability vary with the spatial arrangement of groups. Populations as small as 25 potential breeding groups can be surprisingly resistant to random demographic events, if those groups are highly aggregated in space. Populations as large as 100 potential breeding groups can be impacted by demographic stochasticity, if groups are not aggregated and dispersal of helpers is disrupted. Demographic stochasticity is not expected to affect populations larger than 100 potential breeding groups. Similarly, effects of environmental stochasticity vary with the spatial arrangement of groups. Based on preliminary results of the model and estimates of environmental stochasticity derived from the North Carolina

Sandhills, 250 potential breeding groups will likely withstand effects of environmental stochasticity regardless of their spatial arrangement.

Loss of genetic variation through the process of genetic drift is an inevitable consequence of finite population size. New genetic variation arises through the process of mutation. In large populations, mutation can offset loss through drift and genetic variation is maintained. Just how large a population must be to maintain variation is a difficult question. Currently, researchers recognize that in general, only populations with actual sizes in the thousands, rather than hundreds, can maintain long-term viability and evolutionary potential in the absence of immigration. However, if populations are connected by immigration rates on the order of 1 to 10 migrants per generation (0.5 to 2.5 migrants per year), the genetic variation maintained by these populations is equal to that of one population as large as the sum of the connected populations. Thus, sufficient connectivity among populations can maintain genetic variation and long-term viability for the species.

RECOVERY GOAL

The ultimate recovery goal is species viability. This goal is represented by delisting. Once delisting criteria are met, the size, number, and distribution of populations will be sufficient to counteract threats of demographic, environmental, genetic, and catastrophic stochastic events, thereby maintaining long-term viability for the species as defined by current understanding of these processes. Regions and habitat types currently occupied by the species will be represented to the best of our ability, given habitat limitations.

RECOVERY CRITERIA

Recovery criteria have been formulated using eleven recovery units delineated according to ecoregions. Populations required for recovery are distributed among recovery units to ensure the representation of broad geographic and genetic variation in the species. Viable populations within each recovery unit, to the extent allowed by habitat limitations, are essential to the recovery of the species as a whole.

Population sizes identified in recovery criteria are measured in number of potential breeding groups. A potential breeding group is an adult female and adult male that occupy the same cluster, with or without one or more helpers, whether or not they attempt to nest or successfully fledge young. A traditional measure of population size has been number of active clusters. Potential breeding groups is a better measure of population status, because this is the basis of population dynamics in this species, and number of active clusters can include varying proportions of solitary males and captured clusters. Estimates of all three parameters—number of active clusters, proportion of solitary males, and proportion of captured clusters—are required to support estimates of potential breeding groups.

To assist in the transition between these two measures, we have provided a range of numbers of active clusters considered the likely equivalents of the required number of potential breeding groups. Estimated number of active clusters is likely to be at least 1.1 times the number of potential breeding groups, but it is unlikely to be more than 1.4 times this number. Thus, an estimated 400 to 500 active clusters will be necessary to contain 350 potential breeding groups, depending on the proportions of solitary males and captured clusters and also on the estimated error of the sampling scheme. It is expected that all recovery populations will have sampling in place that is adequate to judge potential breeding groups. If this is not the case, only the highest number of active clusters in the range given can be substituted to meet the required population size.

Delisting

Delisting shall occur when each of the following criteria is met. Rationale for each criterion is given immediately following this list.

Criterion 1. There are 10 populations of red-cockaded woodpeckers that each contain at least 350 potential breeding groups (400 to 500 active clusters), from among the 13 designated primary core populations, and these 10 populations exhibit a stable or increasing trend. A population is considered to exhibit a decreasing trend if a 10 percent decline in the number of potential breeding groups is documented from one year to the next, or if for three consecutive years the number of potential breeding groups declines. If a population does not show decreasing trend, it is stable or increasing. All 13 primary core populations should be managed for maximum size that the habitat will allow.

Criterion 2. There are 10 populations of red-cockaded woodpeckers that each contain at least 250 potential breeding groups (275 to 350 active clusters), from among the 12 designated secondary core populations, and these 10 populations exhibit a stable or increasing trend. All 12 secondary core populations should be managed for maximum size that the habitat will allow.

Criterion 3. There are at least 250 potential breeding groups (275 to 350 active clusters) distributed among designated essential support populations in the South/Central Florida Recovery Unit, and six of these populations including either Big Cypress National Preserve or Ocala National Forest exhibit a minimum population size of 40 potential breeding groups and a stable or increasing trend. Essential support populations in this recovery unit should be managed for maximum population size that the habitat will allow.

Criterion 4. There is one stable or increasing population containing at least 100 potential breeding groups (110 to 140 active clusters) in northeastern North Carolina and southeastern Virginia.

Criterion 5. For each of 12 primary core populations and 13 secondary core populations, and for 6 essential support populations in the South/Central Florida Recovery Unit,

responsible management agencies shall provide (1) a habitat management plan that is adequate to sustain the population and emphasizes frequent prescribed burning, and (2) a plan for continued population monitoring.

Rationale for Delisting Criteria

Criterion 1. A population size of 350 potential breeding groups is considered highly robust to threats from environmental stochasticity as well as inbreeding and demographic stochasticity. It is the lowest of current estimated minimum sizes necessary to offset losses of genetic variation through genetic drift. Ten of 13 are required for delisting because it is recognized that at any given time, two primary core populations may be suffering hurricane impacts. Thirteen primary core populations are designated because of available habitat and because this number, together with 12 secondary core populations (below), may serve to facilitate natural dispersal among populations and maximize retention of genetic variability. Primary and secondary core populations provide for the conservation of the species within each major physiographic unit in which it currently exists, with the exception of South/Central Florida. This unit is represented by several, smaller, essential support populations (below).

Criterion 2. A population size of 250 potential breeding groups is the minimum size considered robust to environmental stochasticity, and is well above the size necessary to withstand inbreeding and demographic stochasticity. Ten of 12 designated secondary core populations are required for delisting to allow for hurricane impacts.

Criterion 3. This unique habitat type is represented to the extent that available habitat allows. Unique genetic resources are conserved as much as reasonably possible. Because of small size, some of these populations will remain vulnerable to extinction threats and may eventually be lost. The likelihood of extirpation of small populations can be minimized by aggregating territories in space.

Criterion 4. This unique habitat, and genetic resources contained within this population, will be represented at the time of delisting. This population size is midway in estimates of sizes necessary to withstand threats from inbreeding depression and are considered robust to demographic stochasticity if territories are moderately aggregated in space.

Criterion 5. Continued habitat management and population monitoring are necessary to ensure that the species does not again fall to threatened or endangered status.

Downlisting

Downlisting shall occur when each of the following criteria is met. Rationale for each criterion is presented immediately following this list.

Criterion 1. There is at least one stable or increasing population containing at least 250 potential breeding groups (275 to 350 active clusters) in each of the following recovery

units: Sandhills, Mid-Atlantic Coastal Plain, South Atlantic Coastal Plain, East Gulf Coastal Plain, West Gulf Coastal Plain, Upper West Gulf Coastal Plain, and Upper East Gulf Coastal Plain.

Criterion 2. There is at least one stable or increasing population containing at least 100 potential breeding groups (110 to 140 active clusters) in each of the following recovery units: Mid-Atlantic Coastal Plain, South Atlantic Coastal Plain, and East Gulf Coastal Plain.

Criterion 3. There is at least one stable or increasing population containing at least 70 potential breeding groups (75 to 100 active clusters) in each of the following recovery units: Ouachita Mountains, Cumberlands, and Piedmont. In addition, there is one stable or increasing population containing at least 70 potential breeding groups (75 to 100 active clusters) in northeastern North Carolina/southeastern Virginia.

Criterion 4. There are at least four populations each containing at least 40 potential breeding groups (45 to 60 active clusters) on state and/or federal lands in the South/Central Florida Recovery Unit.

Criterion 5. There are habitat management plans in place in each of the above populations identifying management actions sufficient to increase the populations to recovery levels, with special emphasis on frequent prescribed burning during the growing season.

Rationale for Downlisting Criteria

Criterion 1. This population size, 250 potential breeding groups, is sufficient to withstand extinction threats from environmental uncertainty, demographic uncertainty, and inbreeding depression. These seven populations will represent each major recovery unit.

Criterion 2. A second population in these coastal recovery units will decrease the species' vulnerability to hurricanes. The West Gulf Coastal Plain is excluded because there are no candidate populations there. The lower size, 100 potential breeding groups, is considered sufficient to withstand threats from demographic uncertainty and inbreeding depression, and is much more quickly attained than 250 potential breeding groups thought necessary to withstand environmental stochasticity.

Criterion 3. Populations in these special habitats will have a decent foothold on survival at the time of downlisting. This population size is midway in estimates of sizes necessary to withstand threats from inbreeding depression and is considered robust to demographic stochasticity if territories are moderately aggregated in space.

Criterion 4. This unique region will be represented at the time of downlisting. Forty active clusters is at the lower end of estimates of sizes necessary to withstand inbreeding

depression and are considered robust to demographic stochasticity if territories are highly aggregated in space.

Criterion 5. These habitat management plans are necessary to ensure progress toward delisting.

ACTIONS NEEDED

The primary actions needed to accomplish the ultimate (delisting) and interim (downlisting) recovery goals are (1) application of frequent fire to both clusters and foraging habitat, (2) protection and development of large, mature pines throughout the landscape, (3) protection of existing cavities and judicious provisioning of artificial cavities, (4) provision of sufficient recruitment clusters in locations chosen to enhance the spatial arrangement of groups, and (5) restoration of sufficient habitat quality and quantity to support the large populations necessary for recovery.

DATE OF RECOVERY

We estimate that, with full implementation of this recovery plan, red-cockaded woodpeckers will be downlisted by the year 2035 and delisted by 2067.

PART I. INTRODUCTION

1. LISTING

A. REASONS FOR LISTING

The red-cockaded woodpecker was listed as endangered in 1970 (35 Federal Register 16047) and received federal protection with the passage of the Endangered Species Act in 1973. Once a common bird distributed continuously across the southeastern United States, by the time of listing the species had declined to fewer than 10,000 individuals in widely scattered, isolated, and declining populations (Jackson 1971, Ligon *et al.* 1986).

This precipitous decline was caused by an almost complete loss of habitat. Fire-maintained old-growth pine savannahs and woodlands that once dominated the southeast, on which the woodpeckers depend, no longer exist except in a few small patches. Longleaf pine (*Pinus palustris*) ecosystems, of primary importance to red-cockaded woodpeckers, are now among the most endangered systems on earth (Simberloff 1993, Ware *et al.* 1993). Shortleaf (*P. echinata*), loblolly (*P. taeda*), and slash pine (*P. elliottii*) ecosystems, important to red-cockaded woodpeckers outside the range of longleaf, also have suffered severe declines (Smith and Martin 1995).

Loss of the original pine ecosystems was primarily due to intense logging for lumber and agriculture. Logging was especially intense at the turn of the century (Frost 1993, Martin and Boyce 1993, Conner *et al.* in press). Two additional factors resulting in the loss of original pine systems in the 1800's and earlier were exploitation for pine resins and grazing by free-ranging hogs (*Sus scrofa*; Wahlenburg 1946, Frost 1993). In the 1900's, fire suppression and detrimental silvicultural practices had major impacts on primary ecosystem remnants, second-growth forests, and consequently on the status of red-cockaded woodpeckers (Frost 1993, Ware *et al.* 1993, Ligon *et al.* 1986, 1991, Landers *et al.* 1995, Conner *et al.* in press). After initial cutting, longleaf forests suffered a widespread failure to reproduce, at first because of hogs and later because of fire suppression (Wahlenburg 1946, Ware *et al.* 1993). These factors are discussed in more detail below.

Loss of the Original Ecosystems

Southern pine savannahs and open woodlands once dominated the southeastern United States, and may have totaled over 80 million ha (200 million ac) at the time of European colonization (Conner *et al.*, in press). Longleaf pine communities characterized the Atlantic and Gulf coastal regions, and covered an estimated 24 to 37 million ha (60 to 92 million ac; Wahlenburg 1946, Frost 1993, Ware *et al.* 1993, Landers *et al.* 1995). Roughly one quarter of the longleaf communities also supported other pines such as loblolly, shortleaf, slash, and pond pine (*P. serotina*) in various proportions

depending on soil conditions, especially in transitional zones between the coastal plains and other physiographic regions (Frost 1993, Landers *et al.* 1995).

Today, longleaf forests have declined to less than 1.2 million ha (3 million ac; Landers *et al.* 1995), of which roughly 3 percent remains in relatively natural condition (Frost 1993). Little old-growth remains, and virtually no longleaf forest has escaped changes in the natural fire regime (Simberloff 1993, Walker 1999). Shortleaf pine was prevalent outside the range of longleaf, especially on dry slopes and ridges in the Interior Highlands and Oklahoma, and has declined considerably (Landers 1991, Smith and Martin 1995). In the precolonial forests, loblolly was present as a minor component of riparian hardwoods or in association with shortleaf in some upland interior forests (Landers 1991).

Southern pine forests today are very different from precolonial communities not only in extent, but also in species composition, age, and structure (Ware *et al.* 1993, Noel *et al.* 1998). Original pine forests were old, open, contained a two-layered structure of canopy and diverse grass and forb groundcover, and were dominated by longleaf in the coastal plain, longleaf/shortleaf/loblolly in the Piedmont and interior highlands, and slash pine (*P. elliottii* var. *densa*) in south Florida. Much of today's forest is young, dense, and dominated by loblolly pine, with a substantial hardwood component and little or no groundcover (Ware *et al.* 1993).

Original longleaf communities in the Atlantic and Gulf coastal plains were first heavily impacted by exploitation for naval stores and then virtually eliminated by widespread logging and subsequent reproductive failure of longleaf pine (Frost 1993, Ware *et al.* 1993). Naval stores industries harvested pine resin for the production of tar, pitch, and turpentine—commodities in high demand during colonial times. Pine woodlands were logged for lumber and conversion to agricultural fields. Impacts to easily accessible areas began with the arrival of Europeans, but technological developments of the 1800's, such as the copper still, steam power, and especially railroads, dramatically increased the rate and area of loss (Frost 1993). In the late 1800's logging operations moved to the previously inaccessible interior forests of longleaf, shortleaf, and loblolly pines. For over a decade these operations removed a reported 3 to 4 billion board feet per year (Frost 1993), and an estimated 13 billion board feet of longleaf was extracted in 1907 alone (Wahlenburg 1946, Landers *et al.* 1995). This especially intense period of logging from 1870 to 1930 resulted in the loss of virtually all remaining old-growth forest in the southeast (Frost 1993, Martin and Boyce 1993, Conner *et al.*, in press).

A common logging practice before the late 1800's was to leave a fair number of residual trees, including small trees, some of those infected with red heart fungus (*Phellinus pini*), and some that had been boxed for resin production (Wahlenburg 1946, Conner *et al.*, in press). Cavity trees of red-cockaded woodpeckers probably were left in much higher proportion than their numbers, due to the likelihood of red heart infection and the abundant resin coating. These residual pines enabled the red-cockaded woodpeckers to survive the original devastation. Loss of residual trees in the twentieth

century has been a major factor in the decline of woodpecker populations (Costa and Escano 1989, Conner *et al.*, in press; see 2D).

Fire Suppression

Precolonial fire frequencies in the southeast have been estimated at 1 to 3 years for the Atlantic and lower Gulf coastal plains (Stout and Marion 1993, Ware *et al.* 1993, Frost 1998), 4 to 6 years for the Piedmont and upper Gulf coastal plain, and 7 to 25 years for the southern Appalachians and interior highlands (Masters *et al.* 1995, Frost 1998). Fire frequency increases with size of fire compartments, and natural firebreaks in the southeastern coastal plains were rare (Ware *et al.* 1993, Frost 1998). Historically, fires were ignited by Native Americans and by lightning. Lightning was the primary ignition source shaping the evolution of these fire-maintained ecosystems, but Native Americans may have played a substantial role in maintaining them (Delcourt *et al.* 1993, Frost 1993). Such maintenance vanished, of course, as Native Americans were all but eradicated by the diseases and aggression of incoming Europeans. Natural fire frequency also declined as fires were reduced in area because of roads, plowed fields, and other human-made firebreaks (Frost 1993, Ware *et al.* 1993).

Europeans brought their perceptions of fire with them as they colonized North America. In Europe, fire was an integral part of traditional swidden agriculture (i.e., shifting cultivation) and was celebrated by peasants as a source of renewal (Pyne 1998). In contrast, urban intellectuals and authorities viewed fire as a destructive force. This view was rooted in a social context: controlling the use of fire could facilitate control of the populace by discouraging the nomadic system (Pyne 1998). Such socially constructed perceptions of fire impacted natural fire regimes in all of Europe's colonies (Pyne 1998).

In North America, after European settlement and prior to the mid 1800's, farmers burned the woodlands regularly to improve forage for free-ranging livestock. Burning the open woods decreased with the fencing of livestock in the mid to late 1800's (Frost 1993), although many people continued to use fire in agricultural fields well into the 1900's (Martin and Boyce 1993). In the twentieth century, the rise of mechanical and chemical agriculture has replaced fire-based agricultural methods.

Active fire suppression began to be institutionalized in the southeastern United States between 1910 and 1930 (Frost 1993, Ware *et al.* 1993). Several factors influenced its rise. First was the existing bias against fire brought to this continent by European intellectuals (Pyne 1998). Then, in the late 1800's, fire suppression grew in response to the extreme intensity of fires burning the logged-over slash across the entire eastern United States. Fires in pine resin orchards were similarly intense and had been suppressed for some time to protect resin production (Frost 1993). Fire was denounced by many ecologists as detrimental to southern pines rather than an integral or useful component of the natural system. Suppression of fire increased with the rise of pine plantations, a land use which began in the 1930's and 40's and continues to increase today (Martin and Boyce 1993, Stout and Marion 1993, Ware *et al.* 1993).

Fire suppression has severe and numerous impacts on southern pine ecosystems, including changes in tree species composition and forest structure. Longleaf pine cannot reproduce without access to the mineral soil, and will be replaced under fire suppression by other species of pines and hardwoods. Structure changes from two layers, a canopy and a diverse groundcover, to a multi-layered midstory and canopy and little or no groundcover. With increasing hardwood midstory, arthropod communities change in species abundance, species composition, and distribution on the substrate (Collins 1998). Red-cockaded woodpeckers are directly and adversely affected by each of these changes (see 2D and 2E).

Reproduction of longleaf pine has been severely restricted since the precolonial era, first because of the impacts of free-ranging hogs and more recently because of the absence of fire (Wahlenburg 1946, Frost 1993, Ware *et al.* 1993). A short period of reproduction took place after hogs were fenced and before fires were suppressed. Most second-growth longleaf in existence today is 70 to 100 years in age and reproduced naturally during this short period of opportunity (Kelly and Bechtold 1990, Frost 1993, Landers *et al.* 1995). Reproduction of longleaf in the twentieth century has been, and still is, constrained by hardwood midstory developed as a result of fire suppression (Landers *et al.* 1995, Frost 1993, Peet and Allard 1993).

Detrimental Silvicultural Practices

Several silvicultural practices have been detrimental to red-cockaded woodpeckers, including short rotations, clearcutting, and conversion to sub-optimal pine species. Cutting of second-growth longleaf pines began during World War II and continues today. Removal of second-growth longleaf has exceeded growth by over 40 percent, and much of the remaining longleaf is aging without replacement (Landers *et al.* 1995).

The years following World War II also saw the rise of plantation forestry. Plantations of dense slash or loblolly pines covered over 4.9 million ha (12 million ac) by the mid 1960's and over 6.1 million ha (15 million ac) at present (Ware *et al.* 1993). Plantations typically have been under rotations of 35 to 70 years for sawtimber production and 20 to 40 years for pulp production (Conner *et al.*, in press), and industry has continued to shift from logs and poles to pulp (Landers *et al.* 1995). With technological developments such as chainsaws, the practice of leaving 'cull' pines that were infected with red heart fungus or boxed for resin production declined. These two practices—short rotations and the removal of all trees—had substantial negative impacts on the woodpecker populations that remained after the initial logging (Conner *et al.*, in press).

B. CURRENT THREATS

Despite protection under the Endangered Species Act in 1973, populations of red-cockaded woodpeckers continued to decline throughout the 1970's and into the 1980's in all parts of the species' range (Baker 1983, Ligon *et al.* 1986, 1991, Ortego and Lay

1988, Conner and Rudolph 1989, Costa and Escano 1989, James 1991, 1995, Haig *et al.* 1993, Kelly *et al.* 1994). Only one population was reported to be increasing during this time (Hooper *et al.* 1991a). In the 1990's, most populations were stabilized and many have shown increases (USFWS, unpublished). Stabilizing the declines was the result of a new understanding of population dynamics (see 2B) and the use of powerful management tools such as artificial cavities and prescribed burning (see 3B and 3F). Our challenge now is to bring about the widespread increases in population sizes necessary to recover the species.

Primary threats to species viability for red-cockaded woodpeckers all have the same basic cause: lack of suitable habitat. Red-cockaded woodpeckers require open mature pine woodlands and savannahs maintained by frequent fire, and there is very little of this habitat remaining (Lennartz *et al.* 1983b, Frost 1993, Simberloff 1993, Ware *et al.* 1993). On public and private lands, both the quantity and quality of red-cockaded woodpecker habitat are impacted by past and current fire suppression and detrimental silvicultural practices (Ligon *et al.* 1986, 1991, Baker 1995, Cely and Ferral 1995, Masters *et al.* 1995, Conner *et al.*, in press). Serious threats stemming from this lack of suitable habitat include (1) insufficient numbers of cavities and continuing net loss of cavity trees (Costa and Escano 1989, James 1995, Hardesty *et al.* 1995); (2) lack of foraging habitat of adequate quality; (3) habitat fragmentation and its effects on genetic variation, dispersal, and demography (Conner and Rudolph 1991b); and (4) fundamental risks of extinction inherent to critically small populations from random demographic, environmental, genetic, and catastrophic events (Shaffer 1981, 1987).

Fire suppression and exclusion has been and still is a profound threat to red-cockaded woodpecker populations (see 2D, 2G). Hardwood encroachment due to fire suppression has been a leading cause of loss of woodpecker groups on both public and private lands and continues to be a major threat throughout the species' range (Van Balen and Doerr 1978, Hovis and Labisky 1985, Conner and Rudolph 1989, 1991a, Costa and Escano 1989, Loeb *et al.* 1992, Baker 1995, Cely and Ferral 1995, Escano 1995, Masters *et al.* 1995). Moreover, most assessments of the impacts of fire suppression on woodpecker groups has focused primarily on the effects of hardwood midstory within the nesting and roosting habitat (cluster). Recent research indicates that exclusion of fire from foraging habitat has negative impacts as well (James *et al.* 1997, Hardesty *et al.* 1997, Doster and James 1998, James *et al.*, in press, Walters *et al.* 2000). Even if nesting and roosting habitat is frequently burned, hardwood midstory and other fire-intolerant species within the foraging habitat can reduce group size and productivity (James *et al.* 1997, Hardesty *et al.* 1997). Thus, negative effects of fire suppression are more pervasive than previously thought.

Widespread and frequent applications of growing season fire throughout lands managed for red-cockaded woodpeckers are essential to the recovery of the species (Conner and Rudolph 1989, 1991a, Baker 1995, James 1995). Regrettably, there are several major difficulties affecting the increased use of fire across the southeast. These difficulties include lack of funding for both public land management agencies and private landowners; prohibitive smoke regulations; increasing density of human populations and

associated development; proliferation of firebreaks such as roads, fields, and power lines; and perhaps most importantly, the prejudice against fire held by many private citizens and some public land managers. As this prejudice, built by decades of intensive anti-fire publicity, shifts toward acceptance of the natural role of fire and its benefits for resource management and catastrophic fire prevention, smoke regulations and funding constraints may change. Extreme caution is needed, however, in moving from restoration to maintenance burns. Should restoration burns of fuel-heavy forests cause loss of human life or property, public perception will be slow to change.

Logging is a major threat to woodpecker populations on private lands (Cely and Ferral 1995). Loss of mature pines continues on private lands. One study estimated the rate of pine cutting on private lands in a part of South Carolina and Georgia at 4.0 percent per year, a rate much higher than those estimated by similar methods for temperate or tropical rainforest (Pinder *et al.* 1999). Most of this cutting occurred in older, natural stands established during the 1930's and 1940's. As second-growth forests mature, there may well be another episode of substantial forest harvest (Ware *et al.* 1993, Landers *et al.* 1995, Pinder *et al.* 1999). Total area, publicly and privately owned, supporting longleaf pine is still declining, from an estimated 1.53 million ha (3.77 million ac) in 1985 (Kelly and Bechtold 1990) to 1.19 million ha (2.95 million ac) in 1995 (Outcalt and Sheffield 1996). Privately owned lands sustained the greatest losses. Private lands also support the majority of longleaf remaining, although much of it occurs in parcels of less than 20.2 ha (50 ac; Outcalt and Sheffield 1996). A major cause of the decline in longleaf pine is replacement by other pine species after logging (Outcalt and Sheffield 1996). Conversion to pine plantations, begun in the 1940's, is still occurring and plantations now cover over 6.1 million ha (15 million ac; Stout and Marion 1993, Ware *et al.* 1993).

Silvicultural practices on public lands have improved in recent years. Agency responses to legislated protection of red-cockaded woodpeckers include longer rotation times (USFS 1995), increases in the area under protection (USFS 1995), and less conversion of native pines to off-site species. For some public lands, timber removal is being viewed as an important management tool rather than an overriding objective (R. Mann, pers. comm.). Overall, current timber production and conversion to off-site pines are less of a threat to red-cockaded woodpeckers on public lands than earlier this century, although effects of past practices are nearly overwhelming.

As described above (this section and 1A), fire suppression and past timber harvests have resulted in an almost complete loss of habitat for red-cockaded woodpeckers. Species recovery is only possible through habitat restoration (see 2D, 2E, 3F, 3G; James 1995, Smith and Martin 1995). However, restoration of habitat may itself jeopardize red-cockaded woodpeckers, if approached without suitable caution. Clearcutting of off-site pine species to restore longleaf and shortleaf pines can potentially disrupt woodpecker populations (Ferral 1998, F. C James, pers. comm.). Restoration of native pines is best achieved through conversion of patches rather than clearcutting large areas, especially if woodpeckers are using off-site pines for foraging or dispersal (Ferral 1998, see 3G).

One of the primary threats to red-cockaded woodpeckers, stemming from past habitat loss, is a severe bottleneck in the number of pines available as cavity trees (Costa and Escano 1989, Rudolph *et al.* 1990b, Conner *et al.* 1991a, Walters *et al.* 1992a). Red-cockaded woodpeckers require older pines for cavity excavation for two reasons: (1) only older pines have sufficient heartwood to house a cavity at preferred cavity heights (Jackson and Jackson 1986, Clark 1993, Conner *et al.* 1994) and (2) older pines are more likely to be infected with red heart fungus (Wahlenburg 1946, Conner *et al.* 1994), and the presence of the associated heartwood decay substantially reduces the time required for cavity excavation (Conner and Rudolph 1995a, see 2D). Red-cockaded woodpeckers survived the 20th century (although at drastically reduced numbers) because timber harvest practices of the 19th and early 20th century left some relict pines standing. Harvest methods used during the mid 20th century did not follow this practice, and many relict pines were cut during this period. Still, most cavity trees in existence today are survivors of the original removal of the primary forest (Jackson *et al.* 1979b, Rudolph and Conner 1991). These pines are older than the surrounding forest and suffer high rates of mortality due to increased effects of wind, lightning, southern pine beetles (*Dendroctonus frontalis*) and other pests, and natural senescence (Jackson *et al.* 1978, Conner *et al.* 1991a, Conner and Rudolph 1995b, Rudolph and Conner 1995, Watson *et al.* 1995). Because the surrounding forest is much younger in age, few potential cavity trees are available as replacements. As second-growth forests are allowed to age, more potential cavity trees will become available. In the meantime, a net loss of cavity trees threatens current populations (Costa and Escano 1989). Crisis intervention through intensive cavity management (artificial cavities and restrictors; see 3A) is helping to offset cavity loss but the threat will remain until mature and old-growth trees are restored.

A second major impact of habitat loss on the viability of red-cockaded woodpeckers is the resultant fragmented distribution. Fragmentation and isolation have occurred both among groups within a population and among populations, with serious consequences for red-cockaded woodpeckers. Red-cockaded woodpeckers are particularly sensitive to effects of isolation because of the limited dispersal characteristic of cooperative breeders (Walters *et al.* 1988a; see 2B). Fragmentation among populations increases loss of genetic variation and threats from random demographic and environmental events (Walters *et al.* 1988a, Conner and Rudolph 1991b, Hooper and Lennartz 1995; see below and 2C), because the dispersal that can help offset such threats is easily disrupted. Fragmentation and isolation of groups within a population can substantially increase that population's risk of extinction (Crowder *et al.* 1998, Letcher *et al.* 1998). Populations of red-cockaded woodpeckers are surprisingly persistent if the spatial arrangement of groups within the population is tightly clumped. If groups are isolated and dispersal behavior disrupted, risk of population extinction increases (Crowder *et al.* 1998, Letcher *et al.* 1998, see 2C).

Managers have some limited tools to combat effects of fragmentation (e.g., strategic location of recruitment clusters, retention of forest cover, and translocation). More importantly, as populations recover, isolation effects will not be as intensely acute as they are at present, because larger populations have greater resistance to impacts from environmental and demographic threats, greater retention of genetic variation, and thus

greater probability of persistence. However, effects of fragmentation are likely to remain serious threats to population viability throughout the period of recovery.

A third threat to red-cockaded woodpeckers from past habitat loss is lack of suitable foraging habitat. As described above, recent research indicates that optimal foraging habitat is maintained by fire and contains an old-growth or mature pine component (Conner *et al.* 1991b, Hardesty *et al.* 1997, James *et al.* 1997, in press, Walters *et al.* 2000). Restoration of foraging habitat will likely increase red-cockaded woodpecker densities (Walters *et al.* 2000, James *et al.*, in press; see 2E), which in turn will positively influence demography and dispersal. However, the threat to woodpecker populations from low-quality or insufficient foraging habitat is not as immediate as threats from habitat fragmentation and lack of suitable nesting habitat. Fragmentation and lack of nesting habitat are presently limiting populations and are largely responsible for recent declines. Foraging habitat, on the other hand, affects population densities but will probably not become limiting until the other two problems are solved. It is an important concern for long-term viability.

One last identified threat to species viability that stems from habitat loss is the set of risks inherent to critically small populations. These are similar to fragmentation effects, but rather than occurring through isolation, these threats are related to population size. Small populations may be extirpated because of random environmental, demographic, genetic, and catastrophic events (Shaffer 1981, 1987; see 2C). Random environmental events affect an entire population; for example, an exceptionally severe winter that causes high adult mortality. Random demographic events act on individuals within populations; for example, a death due to predation, or a brood consisting of all males. Random genetic events are losses or gains in frequency of any given gene, simply due to chance inheritance. Lastly, catastrophic events, which can affect large as well as small populations, are similar to environmental events but larger in scale. Any of these processes alone or in concert can cause the extirpation of a small population. Such processes will continue to remain threats until population sizes are sufficient to withstand them (Shaffer 1981, 1987, Crowder *et al.* 1998, Letcher *et al.* 1998; see 2C). Catastrophes will continue to threaten even the largest populations in perpetuity, although the species as a whole will not be in danger once enough large populations are established (e.g., Hooper and McAdie 1995).

Other factors unrelated to habitat loss may threaten red-cockaded woodpeckers, but their importance has not yet been determined. Foremost among unevaluated threats are the risks from pesticides and other environmental contaminants. Suburban groups of woodpeckers may be at especially high risk of adverse effects from toxins. Similarly, impacts of exotic species have not yet been assessed. Exotic plants, especially, may be negatively affecting woodpeckers in some parts of their range (e.g., Lippincott 1998).

Unlike many endangered and threatened species, red-cockaded woodpeckers are well studied (see Jackson 1995). Biologists are developing a good understanding of what constitutes optimal habitat for this species. Although some debate may exist as to the best way to restore these conditions, a detailed picture of excellent red-cockaded

woodpecker habitat is emerging. In addition, managers are now equipped with effective tools to stabilize existing populations until sufficient quantity and quality of habitat for self-sustaining populations can be provided (Walters 1991). However, such habitat restoration and interim crisis management requires ample funding and a strong political will (Conner *et al.*, in press). Any weakness in determination or political will, with accompanying changes in law and policy, would constitute an extremely serious threat to the species.

2. GENERAL BIOLOGY AND ECOLOGY

A. TAXONOMY AND SPECIES DESCRIPTION

Red-cockaded woodpeckers are currently recognized as *Picoides borealis*. The species is endemic to the southeastern United States but other members of the genus are found throughout the Americas, Europe, Asia, and Africa. Red-cockaded woodpeckers were first described for science as *Picus borealis*, “le pic boreal”, by the French businessman and amateur naturalist Vieillot (1807). In 1810, unaware of Vieillot’s description, Alexander Wilson described the species as *Picus querulus* because of its distinctive vocalizations (Wilson 1810).

Wilson gave the species the English common name we use today, red-cockaded woodpecker, in reference to the several red feathers of males, located between the black crown and cheek patch, that are briefly displayed when the male is excited. In Wilson’s time, “cockade” was a common term for a ribbon or other ornament worn on a hat as a badge. The cockade is a poor field mark because it is rarely seen in the field, but does identify the sexes of adult birds in the hand.

Red-cockaded woodpeckers are relatively small. Adults measure 20 to 23 cm (8 to 9 in) and weigh roughly 40 to 55 g (1.5 to 1.75 oz; Jackson 1994, Conner *et al.*, in press). They are larger than downy woodpeckers (*P. pubescens*), similar in size to yellow-bellied sapsuckers (*Sphyrapicus varius*), and smaller than other southeastern woodpeckers. Size of red-cockaded woodpeckers varies geographically, with larger birds to the north (Mengel and Jackson 1977). Because of this, Wetmore (1941) considered the birds of peninsular Florida to be a subspecies (*P. b. hylonomus*) which was later recognized by the American Ornithologists’ Union (1957). Mengel and Jackson (1977), however, examined a larger series of specimens and considered the variation in the species to be smoothly clinal with no justification for distinguishing the birds in south Florida from those elsewhere.

Red-cockaded woodpeckers are black and white with a ladder back and large white cheek patches. These cheek patches distinguish red-cockaded woodpeckers from all others in their range. Red-cockadeds are black above with black and white barring on their backs and wings. Their breasts and bellies are white to grayish white with distinctive black spots along the sides of breast changing to bars on the flanks. Central tail feathers are black and outer tail feathers are white with black barring. Adults have black crowns, a narrow white line above the black eye, a heavy black stripe separating

the white cheek from a white throat, and white to grayish or buffy nasal tufts. Bills are black, and legs are gray to black.

Sexes of adult red-cockaded woodpeckers are extremely similar in plumage and generally indistinguishable in the field. In contrast, sexes of juveniles can be distinguished in the field until the first fall molt, because juvenile females have black crowns whereas juvenile males have red crown patches. Sexes of nestlings in the hand often can be distinguished by eight days of age: capital feather tracks, observed through the transparent skin before feather emergence, appear grayish black in females and reddish in males (Jackson 1982).

Juveniles may be distinguished from adults in the field by duller plumage, white flecks often present just above the bill on the forehead, and by diffuse black shading in the white cheek patch. In the hand, red-cockaded woodpeckers can be aged by the relative length and shape of the vestigial tenth primary until this primary is molted in the fall. This primary of juveniles is longer and more rounded than that of adults (Jackson 1979a). Second-year red-cockaded woodpeckers often can be identified because juveniles do not molt their secondaries during their first fall molt, whereas older birds do. As a result, the secondaries of juveniles during the second calendar year appear more worn and brown in contrast to newer black primaries (Jackson 1994).

B. SOCIOBIOLOGY AND COOPERATIVE BREEDING

The Breeding System

Red-cockaded woodpeckers live in groups that share, and jointly defend, all-purpose territories throughout the year. Group living is a characteristic of their cooperative breeding system. Red-cockaded woodpeckers are one of only a handful of bird species found in the United States that exhibit this unusual system. In cooperative breeding systems, some mature adults forego reproduction and instead assist in raising the offspring of others (Emlen 1991). The cooperative breeding system of red-cockaded woodpeckers is well studied, and several recent reviews are available (Walters 1990, 1991, Jackson 1994). In this species, most helpers are males that remain and assist the breeders, who typically are their parents or other close kin, on their natal territory (Ligon 1970, Lennartz and Harlow 1979, Lennartz *et al.* 1987, Walters *et al.* 1988a). A few females become helpers on their natal territories, and a few individuals of each sex disperse to become helpers of unrelated breeders in other groups (Lennartz *et al.* 1987, Walters *et al.* 1988a, DeLotelle and Epting 1992). Helpers are strictly non-breeders (Haig *et al.* 1994b), but participate in incubation, feeding and brooding of nestlings and feeding of fledglings, as well as territory defense, nest defense, and cavity excavation. Groups may contain as many as four helpers, but most groups consist of only a breeding pair with no helpers, or a breeding pair plus one helper. Groups containing more than two helpers are uncommon.

Red-cockaded woodpecker groups are highly cohesive. Each individual has its own roost cavity, but typically group members congregate immediately after emerging from their cavities at dawn, and then move together through their large territories until they return to their cavities at dusk. Much like a primate troop, they visit only a portion of their territory or home range each day, and travel different routes on different days.

Group formation is best understood in terms of alternative life-history tactics practiced by young birds (Walters 1991). Young birds may either disperse in their first year to search for a breeding vacancy, or they may remain on the natal territory and become a helper. The proportion of each sex adopting each strategy varies among populations (Lennartz *et al.* 1987, Walters *et al.* 1988a, DeLotelle and Epting 1992), but dispersal is always the dominant strategy for females whereas both strategies are common among males. A dispersing individual, if it survives, may become a breeder at age one, but many fail to locate a breeding vacancy and exist as floaters at age one, or in a few cases as helpers in a new group (Walters *et al.* 1988a, 1992a). Some dispersing males locate a territory but no mate, and hence are solitary males at age one. Solitary males and floaters, like helpers (see below), may become breeders at subsequent ages.

It is those individuals who choose to remain at home as helpers rather than disperse that are primarily responsible for group formation. Individuals may remain helpers for up to eight years, but most become breeders within a few years (Walters *et al.* 1988a, 1992a). Helpers may become breeders by inheriting breeding status on their natal territory or by dispersing to a nearby territory to fill a breeding vacancy. When helpers move, it is usually to an adjacent territory, and they rarely disperse across more than two territories.

In contrast, individuals of both sexes dispersing in their first year sometimes move long distances, more than 100 km in a few cases (Walters *et al.* 1988b, Conner *et al.* 1997c; Ferral *et al.* 1997). Still, typical dispersal distances of even first-year birds are much lower than in other avian species. The median dispersal distance of females is only two territories from the natal site, and about 90 percent settle 1 to 4 territories from the natal site (Daniels 1997, Daniels and Walters 2000a). Males are even more sedentary, since many of them adopt the helping strategy. About 70 percent of males become breeders on the natal territory or an immediately adjacent one (Daniels 1997).

Once a male acquires a breeding position, by whatever pathway, he almost invariably holds it until his death (Walters *et al.* 1988a). Females, however, regularly practice breeding dispersal: roughly 10 percent of breeding females switch groups between breeding seasons each year (Walters *et al.* 1988a, Daniels and Walters 2000b). Females invariably depart when their sons inherit breeding status on their territory, but usually remain when a helper unrelated to them inherits breeding status. Females also are likely to leave if their mate dies and there are no helpers to assume the breeding vacancy, rather than pair with an immigrant replacement male, although not all do so. This may be a means to avoid young males as mates (Daniels and Walters 2000b, below). Also, young females (age one or two) that experience reproductive failure are likely to move (Daniels and Walters 2000b). Like first-year birds, dispersing adult females occasionally

move very long distances (Walters *et al.* 1988b), but typically move to a neighboring group (Walters *et al.* 1988a, Daniels 1997).

Reproduction

Red-cockaded woodpeckers are highly monogamous. The group produces a single brood, and the breeding male and female within the territory are almost invariably the genetic parents of all offspring (Haig *et al.* 1993, 1994b). There is no evidence that helpers ever sire offspring, and the frequency of extra-pair fertilization involving individuals outside the group is among the lowest yet recorded in birds (Haig *et al.* 1994b).

Typical values of reproductive parameters, and the range of variation among years and populations, are available from several published studies (Lennartz *et al.* 1987, Walters *et al.* 1988a, Walters 1990, DeLotelle and Epting 1992, LaBranche and Walters 1994, DeLotelle *et al.* 1995, James *et al.* 1997) and unpublished reports (North Carolina Sandhills and coastal North Carolina, Walters and Meekins 1997, Walters *et al.* 1997, 1998; Eglin Air Force Base and Apalachicola National Forest, Florida, Hardesty *et al.* 1997). Unless otherwise indicated, values reported below represent a summary of data from these sources.

Not all groups attempt nesting in a given year. On average about 10 percent of the groups do not nest, but this ranges from as low as 3 percent to as high as 21 percent. Groups with young breeders, especially one-year-old males, are especially likely to forego nesting (Walters 1990). If the group does nest, the eggs are usually laid in the most recently completed cavity available, which typically is the breeding male's roost cavity (Conner *et al.* 1998a). If the nest fails, the group may renest. On average about 30 percent of nest failures are followed by a second attempt, but annual variation in the rate of renesting is high. There are records of a group making a third nesting attempt following two failed nests, and of a group attempting a second brood after a successful first nest (LaBranche *et al.* 1994, Schillaci and Smith 1994), but both are exceedingly rare. Equally rare are instances of two nests of a single pair in existence at the same time (Rossell and Britcher 1994, R. Conner *et al.*, in prep., J. Walters, unpublished). It seems that almost any odd variation of the typical reproductive process can occur in rare instances. Other examples include two females residing together within a group and laying clutches synchronously in a common nest, or laying in separate nests. Successful instances of the former, but not the latter, have been observed. Such instances are of theoretical interest because they constitute plural breeding, which is characteristic of more complex types of cooperative breeding systems (Emlen 1991).

Normally, however, one brood is produced as a result of one or perhaps two nesting attempts involving only two parents. Most groups that attempt nesting fledge young, as nest failure rates are low for a species in the temperate zone, although fairly typical for a primary cavity nester (Martin and Liu 1992, Martin 1995). Nest failure rates average about 20 percent, and this is fairly consistent among years and among populations. Nest predation, nest desertion, and loss of nest cavities to cavity

kleptoparasites appear to be the primary causes of nest failure. Failure rate is higher during the egg stage than during the nestling stage, which suggests that nest desertion, rather than nest predation or loss of cavities to kleptoparasites, is the major cause of failure (Ricklefs 1969). The relative frequencies of these three causes of nest loss have never been measured directly, however.

Nest predation rates may be lower than in other cavity nesters because of the protection provided by the resin barrier around the cavity, which clearly interferes with climbing by snakes (Rudolph *et al.* 1990b). The frequency of nest predation may vary regionally, although there is no direct evidence of this. One possibility is that it is higher in areas where most cavities are in species other than longleaf, and thus where the resin barrier is diminished (Conner *et al.* 1998a), for example in Arkansas (Neal 1992).

In contrast to nest predation, nest desertion may be more common than in other cavity nesters because of the complex social system and resulting intense competition for breeding vacancies (see below) characteristic of this species. Lennartz *et al.* (1987) suggested that nest failure is often associated with repeated territorial intrusions by conspecifics, and other forms of social disruption. Immigrants often associate with groups as affiliated floaters or unrelated helpers (Walters *et al.* 1988a). Such individuals are a particularly likely source of social disruption that might cause groups to forego nesting, or fail if they do attempt to nest (DeLotelle and Epting 1992).

The primary cavity kleptoparasites linked to nest failure are red-bellied woodpeckers (*Melanerpes carolinus*), red-headed woodpeckers (*M. erythrocephalus*), eastern bluebirds (*Sialia sialis*), and southern flying squirrels (*Glaucomys volans*). These species are known to usurp nest cavities from red-cockaded woodpeckers and to destroy nests in cavities they usurp. Occasionally, red-headed woodpeckers, red-bellied woodpeckers, and flying squirrels may consume eggs and small nestlings (Jackson 1994).

Although red-cockaded woodpecker groups produce broods fairly reliably, these broods are relatively small. This is because clutch size is modest and, more importantly, because partial brood loss is greater than in other species of primary cavity nesters in the United States (LaBranche and Walters 1994). Most clutches contain 2 to 4 eggs, although the full range is 1 to 5 eggs. Even larger clutches are occasionally reported, but these probably (and in some cases certainly) result from two females laying in the same nest (see above). There is variation among populations in clutch size, with population averages ranging from 2.9 to 3.5 eggs, but there does not appear to be a regular geographic pattern in this variation.

Incubation begins before the clutch is complete, and eggs hatch asynchronously (Jackson 1994). As often occurs in species with asynchronous hatching, partial brood loss occurs soon after hatching. Some reduction in brood size is due to failure of eggs to hatch, but much of it is due to mortality of nestlings within the first few days after hatching. The relative frequencies of these forms of loss are not known precisely, and neither are the mechanisms producing the mortality. Eggs may fail to hatch because they are infertile, but it is likely that some do not hatch because the birds cease incubating

them after the first eggs hatch. It may be that the last young to hatch often starve because they cannot compete with their older siblings for food, or they may even be killed by their siblings. However, it is not even known with certainty that it is the youngest nestlings that die.

Partial brood loss, measured by dividing the number of fledglings by the number of eggs in successful nests, averages about 40 percent. It is, however, highly variable among years and among populations. This is one parameter that appears to exhibit systematic geographic variation. Partial brood loss tends to be higher in coastal populations compared to inland ones, and in southern populations compared to northern ones. Population averages vary from around 30 percent in a northern, inland population (North Carolina Sandhills) to about 50 percent in a southern, coastal population (Eglin Air Force Base in Florida), and 59 percent in central Florida.

The average number of young fledged from successful nests is about two in northern populations. Broods of 1 to 4 are common, and rarely five young are fledged from a single nest. Because some groups do not nest and others fail in their attempts, the average number of young produced per group is about one-half fledgling less, ranging from 1.4 to 1.7 among populations, and from 1.0 to 1.9 among years within populations. Thus one can expect about 1.5 young to be produced per group in northern populations. Productivity in Florida populations typically is somewhat less, due largely to greater partial brood loss. In Florida most groups fledge only one or two young, occasionally three. Annual values range from 0.9 to 1.6, and the typical value for a Florida population is about 1.2 fledglings per group per year.

For the first several days after fledging, the young birds are somewhat reluctant to fly, and spend considerable time perched high up in the pines, clinging to the trunk. Parents and helpers sometimes forage some distance away from the young at this time, but return frequently to feed them. During this initial period, the fledglings often do not return to the cluster with the adults in the evening, but instead roost in the open wherever the adults leave them at the end of the day. The next morning, the adults return and locate the fledglings, and resume feeding them.

By the end of the first week out of the nest, however, the young are much more active, and move with the adults as the group travels through the territory. Frequently fledglings will follow adults closely, and beg loudly for food as the adult forages. They may even displace the adult from a particularly productive foraging location. Fledglings often are highly aggressive toward one another, and clear dominance hierarchies are evident among siblings. Males, which are recognizable from their red crown patches, usually are dominant to females. Most of the aggression consists of a dominant fledgling displacing a subordinate from an adult that is carrying food or foraging. The fledglings gradually begin to obtain food for themselves, but continue to beg for food and squabble with each other for some time. It is not unusual to see young being fed two months after fledging, and young are occasionally seen begging as late as the subsequent winter (Ligon 1970).

The sex ratio among fledglings has been reported as biased toward males in a South Carolina population (Gowaty and Lennartz 1985), biased toward females in a Florida population (Epting and DeLotelle, unpublished), and unbiased (i.e., 1:1) in three North Carolina populations (LaBranche 1992, Walters, unpublished) and another Florida population (Hardesty *et al.* 1997). Examination of data on fledgling sex ratios from other populations across the region reveals similar variability (R. DeLotelle, unpublished). It has been proposed that in some cooperatively breeding birds sex ratios are biased toward the helping sex as an adaptive evolutionary strategy (Gowaty and Lennartz 1985, Emlen *et al.* 1986, Lessells and Avery 1987, Ligon and Ligon 1990). This hypothesis has been referred to as the repayment model (Emlen *et al.* 1986). However, in a close examination of the repayment model, Koenig and Walters (1999) found it unable to account for sex ratios in red-cockaded woodpeckers and that the model itself may not be correct. Also, the model does not explain the observed variation in sex ratios among populations of red-cockaded woodpeckers. Generally the cause of this variation is poorly understood, and in particular the relationship between other demographic factors and fledgling sex ratios remains a mystery. Sex ratio likely will continue to be of theoretical interest, but it has little bearing on management.

As discussed previously, many fledglings remain with the group through their first year and beyond, and become helpers. But even young that disperse in their first year may remain with the group for many months. Some young disperse in late summer, only weeks after fledging. However, most of those who have not yet departed by the onset of cooler weather in autumn remain with their natal group through the winter, and disperse in late February, March or even April. Although both natal and breeding dispersal can occur at any time, the two primary periods during which movement occurs are just before and just after the breeding season.

Helpers contribute substantially to both incubating eggs and feeding young, and their presence increases productivity. Groups with helpers produce more young than groups without helpers, but this is due in part to an association between the presence of helpers and high territory quality, as well as actual contributions of helpers to reproduction. The best estimate of the helper effect, controlling for effects of territory quality, is that productivity is increased by 0.39 fledglings per group per year by the presence of a helper, and by an additional 0.36 fledglings by the presence of a second helper (Heppell *et al.* 1994). For unknown reasons, the usual positive effect of helpers on productivity seems to be lacking in two of the Florida populations (DeLotelle and Epting 1992, Hardesty *et al.* 1997, but see James *et al.* 1997).

The mechanism by which helpers increase productivity is not entirely clear. One might assume that since helpers contribute substantially to feeding, groups with helpers should be able to raise larger broods. Indeed, in some cooperative breeders feeding by helpers results in higher provisioning rates, and reduced partial brood loss. In others, however, feeding by helpers instead results in reduced feeding effort by the breeders, and positive impacts of helpers are due to reduced nest failure rather than reduced partial brood loss (Emlen 1991). The latter scenario may characterize red-cockaded woodpeckers, but the evidence is equivocal. Lennartz *et al.* (1987) reported that higher

productivity by groups with helpers on the Francis Marion National Forest was due to reduced partial brood loss. The extent of partial brood loss also is related to the age of the breeders (see below), however, and breeder age can be confounded with presence of helpers in small data sets. Using a much larger sample, and controlling for the age of the female breeder, Reed and Walters (1996) found that in the North Carolina Sandhills higher productivity of groups with helpers was not due to reduced partial brood loss. Instead, groups with helpers were more likely to attempt nesting, and less likely to fail. Khan (1999) found, for this same population, that feeding by helpers resulted in less feeding by parents rather than more food being delivered to nestlings.

Reproductive success is strongly affected by age in both sexes. Young birds are less successful than old birds, and this is manifested in all components of reproduction. That is, young birds are less likely to attempt nesting, more likely to fail, and suffer more partial brood loss. Productivity of one-year-old birds of both sexes is especially poor, but reduced productivity is evident through age three, and the effect is somewhat stronger in males. Ages 4 to 8 are the peak reproductive years, as productivity is reduced somewhat at ages 9 and beyond in both sexes. This may represent senescence (see below).

Mortality

Data on mortality rates come from the same sources as data on reproduction (see above). Good estimates are available from completely marked populations or subpopulations, and patterns are clear and consistent. For a bird of its size residing in temperate regions, the red-cockaded woodpecker exhibits exceptionally high survival rates. Survival rates of adult male helpers and breeders generally are about 5 percent higher than that of breeding females. There is distinct geographic variation in survival similar to that observed for partial brood loss. Survival rates are about 75 percent for males and 70 percent for females in the northern, inland population in the North Carolina Sandhills, about 80 percent and 75 percent respectively in coastal populations in North Carolina, and 86 percent and 80 percent respectively in central Florida. Such an association between increased survival and reduced fecundity is common in animal life histories. Annual variation in adult survival within populations is sufficiently small that it can largely be attributed to random chance rather than changes in environmental conditions (Walters *et al.* 1988a). This level of variation can have large effects in small populations, however, and it appears that there are occasional poor years in which survival is substantially reduced. Also, some populations are vulnerable to periodic catastrophic mortality due to hurricanes (see 2C).

With survival rates as high as these, it comes as no surprise that some individuals live to old ages. A captive female lived to 17 years (J. Jackson, unpublished), and a male in the North Carolina Sandhills lived to 16 years of age in the wild (J. Carter III, J. Walters and P. Doerr, unpublished). The number of very old birds is less than one might expect, however, because red-cockaded woodpeckers apparently experience senescence. In the North Carolina Sandhills survival rates fall to around 50 percent beginning at age 9 in females and age 11 in males. Survival of one-year-old males is also reduced, but only

if they are breeders: helper males of age one have typical high survival rates. Survival is fairly constant at ages 1 to 10 in males, and 1 to 8 in females.

Survival during the first year is more prone to underestimation than survival at subsequent ages, due to the greater possibility of dispersal out of the sampling area. Nevertheless, it is quite clear that survival rates are much lower during the first year than thereafter. In three North Carolina populations, survival of males during the first year ranges from 46 percent to 57 percent, and of females from 36 percent to 45 percent. Within a population, survival of males is 10 to 15 percent higher than survival of females. It is not clear whether geographic variation in survival during the first year exists, although there is some evidence that survival is higher in Florida (DeLotelle and Epting 1992). Survival during the first year is affected by the proportion of individuals dispersing rather than remaining as helpers (dispersing lowers survival), and by the number of available breeding vacancies (survival improves as the number of vacancies increases), as well as by the physical environment. This makes it more difficult to detect geographic variation.

Differences between age-sex classes suggest that dispersal is associated with reduced survival. By regressing survival against the proportion of birds dispersing among various categories of females, Daniels and Walters (2000b) estimated the mortality cost of movement for breeding females in the North Carolina Sandhills at 33 percent. That is, dispersal between breeding seasons adds another 33 percent to the probability of mortality above what is expected for sedentary birds. Specifically, the expected survival rate for females that do not move is 74 percent, whereas that for females that do move is 41 percent. This is a surprisingly high cost, given the short distances that most individuals move. This result may reflect the intensity of competition for breeding vacancies, the benefits of belonging to a group, or perhaps the benefits of ready access to a suitable roost cavity.

Overall the mortality pattern is fairly typical of cooperatively breeding avian species. It is characterized by relatively low survival during the first year, especially of dispersers; relatively high survival of breeders and helpers; and senescence at the end of the life span. Compared to non-cooperative species, survival of both juveniles and adults is high, and the life span is long.

Population Dynamics

The population dynamics of the red-cockaded woodpecker are intimately related to the species' unusual social system (Walters 1990, 1991). In demographic terms, population dynamics are strongly affected by the presence of a large class of non-breeding adults, helpers. Helpers provide a pool of replacement breeders in addition to young of the year, and thereby act as a buffer between mortality and productivity in regulating population size. That is, the number of breeding groups in one year is not strongly affected by either productivity or mortality in the previous year. Instead, the size of the helper class is affected by these variables, while the number of potential breeding groups remains remarkably constant. If mortality exceeds productivity, the number of

helpers will decrease, because the number of replacement breeders drawn from the helper class will exceed the number of fledglings recruited into it. If productivity exceeds mortality, the opposite will occur, and the number of helpers will increase. Therefore average group size is an important indicator of population health, as it indicates the potential to maintain the size of the breeding population in the face of fluctuations in mortality and productivity. Of course the strength of the buffering effect of helpers depends on the size of the helper class. In small populations the number of helpers may be so few that poor survival or reproduction can have a direct, negative effect on the size of the breeding population (Lennartz and Heckel 1987, DeLotelle *et al.* 1995).

In evolutionary terms, adoption of the helping strategy is closely linked to patterns of territory occupancy (Walters 1990, 1991). Remaining on the natal territory as a helper can be viewed as a strategy, involving delayed reproduction and dispersal, and altered dispersal behavior, to acquire a breeding position. Helpers stay at home and wait for a breeding vacancy to arise in their vicinity, either on the natal territory or a neighboring one (Walters *et al.* 1992b). This strategy is thought to be an effective one when competition for breeding vacancies is intense (Zack and Rabenold 1989). Further, the intense competition for breeding vacancies that characterizes cooperative breeders is thought to result from unusually large variation in territory quality (Stacey and Ligon 1991, Emlen 1991, Koenig *et al.* 1992).

In red-cockaded woodpeckers, variation in territory quality is related to the presence of cavities. Because cavities take so long to construct, an individual does better to acquire a breeding position on an existing territory containing suitable cavities than to occupy vacant habitat and construct new cavities (Walters 1991, Walters *et al.* 1992a, Conner and Rudolph 1995a). Thus habitat lacking suitable cavities is poor quality, and habitat with existing, suitable cavities is high quality. The birds ignore poor quality habitat, even though they could excavate cavities and then reproduce successfully there, and compete intensely for openings in high quality habitat. When artificial cavities are added to unoccupied but otherwise suitable habitat, it immediately becomes high quality habitat, and is quickly occupied (Copeyon *et al.* 1991, Walters *et al.* 1992a).

The implication of this view of population dynamics is that the breeding population size (usually measured as the number of potential breeding groups) is determined by the number of high quality territories, which depends on the number and distribution of suitable cavities. This is consistent with the behavior of populations during the species' decline (Walters 1991), as well as with recent increases in some populations under new management. The dominant feature in population declines has been gradual abandonment of territories rather than poor survival or reproduction. In many cases it is clear that territory abandonment was related to loss of cavities to tree death or cavity enlargement, or to encroachment by hardwood midstory (Jackson 1978b, Van Balen and Doerr 1978, Conner and Rudolph 1989, Costa and Escano 1989). With so many threats to cavities, it was easy to lose territories, and thus populations declined, despite the continued presence of helpers and good productivity on those territories that remained suitable. Often territories are occupied by an unpaired male for a period prior to abandonment, so that response to loss of cavities and other adverse events is delayed

(Jackson 1994). This may be because once territories deteriorate, young birds no longer remain as helpers and females no longer consider them acceptable, but the breeding male refuses to leave. The territory is no longer acceptable to dispersing males, however, so once the original breeding male dies, which may be many years later, the territory is finally abandoned.

New groups on new territories arise by two processes, pioneering and budding (Hooper 1983). Pioneering is the occupation of vacant habitat by construction of a new cavity tree cluster, which according to the view of population dynamics just presented, is expected to be rare. Budding is the splitting of a territory, and the cavity tree cluster within it, into two. Budding is common in many other cooperative breeders, and might be expected to be more common than pioneering in red-cockaded woodpeckers, since the new territory contains cavities from the outset.

The available data indicate that budding indeed is more common than pioneering, and that pioneering is quite rare. In the North Carolina Sandhills, the observed rate of pioneering over 16 years is one event per 1572 existing groups per year, and in Croatan National Forest in coastal North Carolina, over 7 years it is one event per 332 existing groups per year (J. Walters, unpublished). These translate into population growth rates of 0.06 percent and 0.3 percent per year. However, at nearby Marine Corps Base Camp LeJeune, the rate of pioneering over 10 years has been one event per 46 existing groups per year, a population growth rate of 1.5 percent per year (J. Walters, unpublished). During these same periods, rates of population growth through budding have been 0.6 percent, 2.1 percent and 0.6 percent for the Sandhills, Croatan, and LeJeune respectively. Combining budding and pioneering, growth rates are 0.7 percent, 2.4 percent and 2.2 percent per year respectively. During the years when the Sandhills population was declining (1980 to 1984) the growth rate through these processes was 0.1 percent per year, whereas over the subsequent years, when the population was stable, it was 0.9 percent. A population growth rate of 10 percent per year through these processes was reported for the Francis Marion National Forest (Hooper *et al.* 1991a). In this case pioneering and budding events were inferred rather than directly observed, unlike in North Carolina, and it is possible that the rate of population growth was overestimated. Still, this study suggests that the rate of population growth through budding and pioneering potentially can be substantially greater than what has been observed in North Carolina.

Why the rates of budding and pioneering vary so much is a mystery. It appears from the North Carolina data that rates may be higher in small populations (Croatan, LeJeune) than in large ones (Sandhills), but this is inconsistent with the data from the Francis Marion. Another interpretation is that the rates are higher where turnover of breeders is less, and thus opportunities to replace deceased breeders are fewer. A third hypothesis is that budding and pioneering are stimulated by burning specifically, or habitat improvement generally. This is consistent with the North Carolina data in that rates have been higher in recent years in the Sandhills and LeJeune, following reintroduction of growing season fire, and lower in the last several years on Croatan, since burning during the growing season there has ceased. A fourth hypothesis is that

conditions for population growth may be more favorable in flatwoods habitat than in sandhills habitat.

Rates of budding and pioneering may vary for unknown reasons, but it is clear that they are almost always quite low. These rates were too low to counter losses of territories during the 1970's and 1980's when populations were declining, and they limit the potential for recovery currently, even if losses of territories can be prevented. Thus it is easy to understand why, until the advent of artificial cavity construction, populations generally have been stable or declining rather than increasing.

Understanding that population size is determined by the number of territories with suitable cavities makes designing management to increase populations straightforward (Copeyon *et al.* 1991, Walters 1991). To prevent loss of occupied territories, existing cavity trees should be protected, so that a sufficient number of suitable ones are maintained at all times. This can involve eliminating encroaching hardwoods, protecting cavities with restrictors or replacing lost cavities with artificial ones. To increase the number of suitable territories, cavities can be added in unoccupied habitat, such as abandoned territories with existing cavities and completely vacant areas. In theory it might be possible to rehabilitate abandoned territories by placing restrictors on existing cavities or eliminating hardwoods. In practice, however, only recently abandoned territories seem to be reoccupied without the addition of new cavities (Doerr *et al.* 1989). This may be because cavities deteriorate if unused for long periods. Therefore, for both abandoned territories and vacant habitat, usually the only effective means to create a suitable territory is to construct new artificial cavities in open pine habitat.

Where a management strategy based on maintaining and creating suitable territories has been followed, it has been effective in increasing populations. There have been successes at Eglin Air Force Base in Florida (Hardesty *et al.* 1997, J. Walters *et al.*, unpublished), Croatan National Forest and Camp LeJeune in North Carolina (J. Walters *et al.*, unpublished), Fort Stewart in Georgia (T. Beaty, unpublished), Fort Benning (M. Barron, unpublished), Carolina Sandhills National Wildlife Refuge (S. Lanier, unpublished), Piedmont National Wildlife Refuge (R. Shell, unpublished), and Bienville National Forest in Mississippi (D. Elsen). Rates of population increase are similar across sites, suggesting that a rate of increase of 10 percent per year is perhaps the best that can be achieved (without resorting to translocation). It may be that the pool of new breeders (i.e., helpers, floaters, and first-year birds) generally is not large enough to permit higher rates of increase.

The current understanding of population dynamics suggests not only that management designed to increase the number of suitable territories will be effective, but also that management designed instead to increase productivity and survival will be ineffective in most circumstances. Thus measures designed to thwart nest predators, prevent cavity kleptoparasitism (except to prevent cavity enlargement), or eliminate predators of fledglings and adults often will be ineffective in promoting population growth. Such measures may be necessary, however, in intensively managed, extremely small populations where every individual is critically important. The population at the

Savannah River Site provides the best example of successful, intensive management of a small population (Haig *et al.* 1993, Franzreb 1997).

Like so many other characteristic traits of this species, the origin of its complex social system and unusual population dynamics can be traced back to its most unique feature, excavation of cavities for roosting and nesting in live pine trees. The understanding of these relationships that has been achieved is cause for optimism about the future of the species. Unlike for so many other species, it appears that our understanding of the species' biology is sufficient to construct a management strategy likely to produce recovery, and results to date support this supposition. Ability to implement this strategy is now the key to recovery.

C. POPULATION AND SPECIES VIABILITY

A viable species is one that can reasonably be expected to avoid extinction over a long period of time. Similarly, a viable population is one that is self-sustaining over a long period. For any endangered species, achieving species viability is the ultimate conservation goal, and the ultimate objective of a recovery plan such as this one. How species viability relates to population viability is dependent on population structure. Species viability may be achieved by maintaining a number of independent viable populations. Alternatively, species viability may be achieved by maintaining a network of interacting populations, none of which are viable on their own. We conclude that the appropriate strategy for red-cockaded woodpeckers is to maintain a number of independent viable populations. Here we discuss information about population structure that led us to this conclusion, and then how population viability is best achieved.

Population Structure

Given the historic distribution of its habitat and comments by early naturalists about its abundance, it is highly likely that red-cockaded woodpeckers originally were distributed fairly continuously over broad areas. Since the birds are so sedentary (see 2B), one presumes that originally there may have been considerable genetic substructure within populations, but that distinct, genetic population boundaries were lacking. That is, genetic similarity probably changed gradually with distance, rather than suddenly at population boundaries. In fact, it may have been difficult to even delineate distinct populations.

Such is not the case currently. Now the species is distributed largely as distinct populations, with large gaps of unoccupied habitat between them. Many of these populations are quite small, and only a few are of more than modest size (see map insert and Appendix 1). Typical dispersal distances of both sexes are sufficiently short to maintain genetic substructure within populations even under current conditions. Daniels and Walters (2000a) found that an individual's close relatives are highly concentrated within three territories of the natal site. Thus one can expect genetic similarity to change

with distance within populations, as opposed to the uniform structure that occurs when mating is random within populations.

Although this species is highly sedentary compared to other birds, some individuals move long distances (Walters *et al.* 1988a). There is sufficient documentation (Walters *et al.* 1988b, Conner *et al.* 1997c, Ferral *et al.* 1997, R. Costa, unpublished) to conclude that long-distance movements between populations are rare, but regular, events, and that the birds can move through seemingly inhospitable habitat. It appears that movement from small populations into large ones is much more common than the reverse. Because of this, and the rarity of such movements, they are of little consequence demographically; that is, their contribution to sustaining populations is trivial. However, they are frequent enough to be important genetically, and may function to maintain genetic variability within populations. Producing immigrants that contribute to this function may be one of the primary purposes that small support populations serve.

The most reasonable conclusion, based on current information, is that demographically, populations of red-cockaded woodpeckers function as closed populations. That is, their persistence depends totally on within-population demography, and not at all on exchange between populations. Thus red-cockaded woodpeckers do not exhibit any of the various types of metapopulation structure (Stith *et al.* 1996). Local extinction followed by natural recolonization from another population is extremely unlikely for this species. (The event closest to natural recolonization was the appearance of a male from the Savannah River Site within a recruitment cluster on Fort Gordon, two years after the Fort Gordon population was extirpated. This dispersal event would not have resulted in the formation of a breeding pair without the use of translocation.)

Further, immigration rates are too low for one population to rescue another from extinction as occurs in another cooperatively breeding woodpecker, the acorn woodpecker (*M. formicivorous*; Stacey and Taper 1992). Neither are immigration rates high enough to enable source-sink relationships between populations. However, in areas of low density (e.g., northeastern North Carolina), widely scattered groups considerable distances apart may function as a single population. Dispersal distances are longer when population density is lower (Daniels 1997), apparently because the distance moved is a function primarily of the number of groups encountered rather than of habitat, mortality or speed of movement. Thus migration between two sizeable populations only 24.2 km (15 mi) apart may be rare (e.g., only one movement between the Camp LeJeune and Croatan National Forest populations in North Carolina over 11 years), whereas two groups 24.2 km (15 mi) apart in an area of low density (e.g., only one other group between them) may exchange individuals regularly.

Red-cockaded woodpecker populations should not be viewed as closed genetically, however. Nearly all probably experience some immigration, much of it from smaller support populations. Rates of immigration and genetic relationships between populations are not well enough known to determine precisely the rate of gene flow, nor its effect on genetic variability within populations. All that can be said is that the

existence of gene flow needs to be considered when evaluating the genetic viability of populations (see below).

There are, however, both allozyme (Stangel *et al.* 1992, Stangel and Dixon 1995) and random amplified polymorphic DNA (RAPD) data (Haig *et al.* 1994a, 1996) available that reveal general genetic relationships between populations. These data indicate that most (93 percent, Haig *et al.* 1994a) genetic variation occurs among individuals within populations. Genetic differences between populations increase somewhat with geographic distance, but there is little geographic structure to genetic variability. Genetic differences between populations are greater than is typical of birds, but equivalent to those in other endangered birds. However, populations do not exhibit unique alleles. Some small populations exhibit reduced heterozygosity, but not all do, and generally there is no consistent relationship between population size and genetic variability (Stangel and Dixon 1995). All of this information is consistent with recent isolation of populations in a formerly continuously distributed species, with low levels of gene flow between populations. Populations probably are diverging genetically and losing variability currently, but isolation evidently is too recent for them to differ much yet.

Threats to Population Viability

Information on population structure indicates that the best approach to viability is to manage for independent populations that are individually viable, with appropriate recognition of low levels of gene flow between populations. To assess population viability, generally four threats are considered: (1) demographic stochasticity, (2) environmental stochasticity, (3) catastrophes and (4) genetic drift and inbreeding (Shaffer 1981, 1987). All four threats must be adequately addressed to ensure viability. Here we examine each threat, treating demographic stochasticity and environmental stochasticity together as demographic considerations, and catastrophes and genetic concerns as separate issues. In the previous recovery plan (USFWS 1985) only catastrophes and genetic factors were considered.

Demographic Considerations

Demographic stochasticity refers to effects of random events on the reproduction and survival of individuals, whereas environmental stochasticity refers to effects of unpredictable events that alter vital rates. For example, if every individual has a 50 percent probability of annual survival, in a population of 20 individuals 10 will not die each year. Instead some years by chance nine will die, in others eleven and so forth. This is demographic stochasticity, which is analogous to sampling error. It may be that in years with severe winters the probability of survival is only 30 percent, whereas in years with mild winters it is 70 percent. This is an example of environmental stochasticity.

Demographic stochasticity is inevitable, but is usually considered to be a threat only to small populations, i.e., less than 50 individuals (Meffe and Carroll 1997).

Environmental stochasticity varies widely in strength, depending on the species and the nature of its interactions with its environment. Viability in the face of these threats usually is assessed by incorporating them in simulations of population dynamics, and determining the probability of extinction over long time periods of populations of various sizes. The chief obstacle to a comprehensive viability analysis previously has been lack of a suitable population model. Standard, simple population models do not incorporate the social complexity of the species, notably the buffering effect of the large, nonbreeding helper class (see 2B). These complexities can be handled by stage-based matrix models (Caswell 1989, McDonald and Caswell 1992). Application of these models to red-cockaded woodpeckers has produced important insights about population behavior and management (Heppell *et al.* 1994, Maguire *et al.* 1995). But even these models do not incorporate critically important spatial dynamics resulting from helpers filling breeding vacancies only on or very near their natal territory. A model that assumes that nonbreeders fill breeding vacancies randomly within the population cannot be expected to portray population dynamics accurately enough to perform viability analysis.

The advent of spatially-explicit, individual-based simulation models in ecology provides a tool capable of handling the complex population dynamics of red-cockaded woodpeckers (DeAngelis and Gross 1992, Judson 1994, Dunning *et al.* 1995). These models are not without their faults, a notable one being the large number of parameters that must be accurately estimated if model results are to be trusted (Conroy *et al.* 1995). A spatially-explicit, individual-based model of the population dynamics of red-cockaded woodpeckers has been developed by Letcher *et al.* (1998), using data from the North Carolina Sandhills.

Letcher *et al.* (1998) used their model to assess effects of demographic stochasticity on populations of various sizes and spatial distributions. Their most notable result was the strong effect of spatial structure on viability. If territories were highly clumped, populations of as few as 25 groups were remarkably persistent, whereas if territories were scattered, populations as large as 169 groups declined. New group formation through budding and pioneering (see 2B), which was not incorporated in the analysis, would presumably be sufficient to counter the small declines experienced by the largest populations. Still, the model predicts that demographic stochasticity will be a threat to populations as large as 100 groups if spatial structure is poor, but will not be a threat to populations as small as 25 groups if spatial structure is favorable. Recent analyses indicate that even smaller populations, as small as 10 groups, can be remarkably persistent if the territories are maximally clumped (Crowder *et al.* 1998). These model results are consistent with empirical evidence. Across the range it seems that small aggregates of groups persist surprisingly well, whereas small, low-density populations always seem to decline. Even in somewhat larger populations, loss of isolated groups is a problem (Conner and Rudolph 1991b).

We conclude that demographic stochasticity is as usual a threat only to small populations. However, the threshold of vulnerability varies considerably with spatial structure. Vulnerable populations may be twice the typical size, or half the typical size,

depending on the configuration of the population. It certainly is possible to avoid this threat for populations as small as 25 groups, and it may be possible to avoid it for populations of only 10 groups. Managers therefore should strive to aggregate their populations, and to avoid isolation of groups, where isolation is defined as being beyond the dispersal range of helpers. Based on data from the North Carolina Sandhills (Walters *et al.* 1988a, Daniels 1997), 3.2 km (2 mi) appears to be a reasonable standard to use for the maximum dispersal range of helpers (less than 10 percent of helpers [17 of 240] dispersed more than 3.2 km [2 mi]; Daniels 1997). This maximum dispersal distance refers to habitat that contains no barriers to dispersal. The ideal spatial configuration is one in which every group is within dispersal range of helpers from several other groups.

Evaluating environmental stochasticity is more difficult. Letcher *et al.*'s (1998) model is suitable for this purpose, but accuracy of results will depend not only on the validity of the model, but also on estimates of the magnitude of stochasticity. Typically stochasticity is incorporated as annual variation, and therefore the appropriate variance of each demographic parameter must be determined. It is quite clear from available data that annual variation in productivity is considerable, but annual variation in mortality appears to be fairly small (Walters *et al.* 1988a).

Preliminary analyses of population viability incorporating environmental as well as demographic stochasticity have recently been completed using the model developed by Letcher *et al.* (1998). In these analyses, the magnitude of environmental stochasticity was estimated from observed annual variation in the North Carolina Sandhills population, and annual variation in productivity, adult survival, and fledgling survival was incorporated (Crowder *et al.* 1998). Budding was incorporated into the simulations as well. These results suggest that populations of 100 or fewer groups are vulnerable to extinction, even when territories are maximally clumped. Populations of 250 or more groups are not vulnerable to environmental stochasticity, according to these simulations, even if territories are not highly clumped. Viability of populations between 100 and 250 groups depends on spatial configuration as well as population size, although this has not yet been analyzed in detail.

Clearly, more analyses are necessary before a more precise viability criterion can be defined, but results at hand permit some important conclusions. First, as expected, populations must be considerably larger to avoid the threat of environmental stochasticity than they need be to avoid the threat of demographic stochasticity. Second, the population sizes necessary to achieve viability against these two demographic threats are much smaller than is typical. This is an intuitive result, since the presence of helpers can be expected to dampen oscillations in the breeding population caused by variation in productivity and breeder survival. Years of poor productivity, or low breeder survival, will lead to a reduction in the size of the helper class rather than a reduced number of potential breeding groups. Third, the level of assistance, in the form of translocated birds, required to avoid extinction of small populations may be low enough to be feasible. Fourth, spatial configuration becomes increasingly important to viability as populations become smaller.

It is encouraging that population sizes required to avoid demographic threats to viability fall within a range that is achievable. Producing populations of two thousand groups, were that required, would be inconceivable. Managing to produce populations of 250 or more potential breeding groups with a favorable spatial structure, on the other hand, is a realistic goal. Indeed a few populations already match this description.

Genetic Considerations

There are two genetic threats to population viability. The first, inbreeding depression, threatens only small populations, whereas the second, genetic drift, can threaten even large populations (reviewed in Lande 1995). Inbreeding depression reduces the survival and productivity of individuals, and results from the segregation of partially recessive, deleterious alleles. The resulting negative effect on population dynamics increases vulnerability to extinction. The amount of inbreeding depression depends on the rate of inbreeding and the opportunity for selection to purge recessive lethal and semilethal mutations (Lande 1995). Genetic drift results in the loss of genetic variation, which may reduce a species' ability to adapt and persist in a changing environment, and thereby its viability over long time periods. The rate of loss is inversely related to population size and mutation rate, and viability is achieved when the population size is large enough that loss to drift is in equilibrium with gain from mutation.

The red-cockaded woodpecker is one of the few species for which inbreeding depression has been demonstrated in wild populations, as opposed to assumed from theoretical considerations. In the North Carolina Sandhills, productivity of both closely related (i.e., coefficient of relationship greater than 0.125) pairs and their inbred progeny is substantially lower than that of unrelated pairs and their progeny (Daniels and Walters 2000a). This is due to both reduced hatching rates of eggs and reduced survival of fledglings to age one year. These are precisely the sort of traits one expects to be affected by segregation of partially recessive, deleterious alleles, and in fact reduced hatching rate is the classical manifestation of inbreeding depression in domestic birds (Daniels and Walters 2000a).

Although inbreeding depression is clearly a threat to red-cockaded woodpecker populations, its effects may not yet be evident due to the recent nature of reductions in population size. The available genetic data indicate that most small populations do not yet exhibit high levels of homozygosity (see above). Furthermore, Stangel and Dixon (1995) found no evidence that small populations were experiencing increased morphological variability. They examined fluctuating asymmetries of paired characters, which are often used as an indicator of developmental stability (Leary and Allendorf 1989). Developmental instabilities are thought to be one of the manifestations of inbreeding depression.

Although it appears that there has not yet been sufficient time for the various manifestations of inbreeding depression to become prevalent, they can be expected to increase in the near future in populations that remain small and isolated. Franklin (1980)

suggested that populations with an effective size of 50 individuals or less would be vulnerable to inbreeding effects. Since the red-cockaded woodpecker can be characterized as a species in which large populations have been reduced suddenly to small size, it is reasonable to apply this standard to this species. That is, it is unlikely that previous selection has already purged recessive alleles from red-cockaded woodpecker populations. Instead, this species probably is quite vulnerable to this threat.

Effective size refers to an idealized population in which individuals mate randomly and all contribute equally to reproduction. In this hypothetical ideal population, all individuals pass on an equal number of their genes to subsequent generations. Effective size is a theoretical standard used to estimate the retention and loss of genetic variation in a real population. The effective population size itself is never measured directly; it is calculated using formulas based on genetic theory and demographic data collected from real populations.

The actual population size is almost always higher than the effective size, because several characteristics of animals and populations act to make the genetic contribution of individuals to subsequent generations unequal. For example, some pairs or individuals may consistently produce more offspring than others, and some individuals live longer than others. It is mainly this variation in reproductive success that makes effective size less than actual size.

Thus, it is possible to calculate the effective size of a population if its demography is known. Such calculations indicate that for red-cockaded woodpeckers, the actual population size needed to achieve an effective size of 50 individuals is 31 to 39 potential breeding groups, depending on the details of the demography of particular populations (Reed *et al.* 1988, 1993, D. Heckel and M. Lennartz, unpublished). According to Franklin's (1980) suggestion that an effective size of 50 is necessary to withstand threats from inbreeding depression, stable or increasing populations of 40 or more groups are not threatened by inbreeding depression.

Daniels *et al.* (in press) came to a fairly similar conclusion by using the spatially explicit model developed by Letcher *et al.* (1998). They estimated inbreeding levels over time in red-cockaded woodpecker populations of various sizes and rates of immigration. In their simulations, mean inbreeding increased rapidly in very small, declining populations with no immigration, but remained tolerably low in closed, stable populations of 100 active territories. Moderately high levels of immigration were required to stabilize small declining populations and maintain reasonable inbreeding levels (kinship coefficients less than 0.10). That is, inbreeding depression is not expected to affect populations that are receiving 2 or more migrants per year. In the absence of immigration, Daniels *et al.* (in press) found that a stable population of 50 to 100 or more breeding groups was necessary to avoid inbreeding depression. Thus, based on the work by Daniels *et al.* (in press) as well as Franklin's (1980) initial suggestion, we conclude that stable or increasing populations of at least 40, and possibly as many as 100 potential breeding groups—or an immigration rate of 2 or more migrants per year—are required to protect against inbreeding depression.

The population size necessary to avoid loss of genetic variation due to genetic drift, however, is much larger. Franklin (1980) first proposed that an effective size of 500 individuals would allow maintenance of long-term viability, because loss of genetic variation from drift would be offset by the creation of new variation through natural mutation. Recently, however, this number has been a topic of some debate (Lande 1995, Franklin and Frankham 1998, Lynch and Lande 1998, Allendorf and Ryman, in press). Lande (1995) indicated that only populations with an effective size of over 5000 individuals can be expected to maintain viability in the absence of immigration, because not all mutations are beneficial. Others argue that an effective population size of 500 to 1000 individuals is sufficient (Franklin and Frankham 1998). At issue is the potential effects of harmful mutations: Franklin and Frankham (1998) consider these effects negligible, but others have suggested that slightly deleterious mutations are capable of causing population extinction even at effective sizes of several hundreds (Lande 1994, Lynch *et al.* 1995, Lynch and Lande 1998). The debate will likely continue, but a reasonable conclusion is that only populations with actual sizes in the thousands, rather than hundreds, can maintain long-term viability and evolutionary potential in the absence of immigration (Allendorf and Ryman, in press).

Thus, without immigration, no recovery population will be large enough to avoid loss of genetic variability through drift. One practical way to reduce this threat is to promote immigration, both natural (from support and other core populations) and artificial (from translocation). Sufficient connectivity among populations, in the order of 1 to 10 migrants per generation in each direction (0.25 to 2.5 migrants per year), can maintain genetic variation and long-term viability for the species (Mills and Allendorf 1996). Populations connected by this level of immigration maintain genetic variation equal to that of one population as large as the sum of the connected populations (F. Allendorf, pers. comm.). As populations increase, natural dispersal among them will likely increase, but determining actual rates of natural immigration is a critical research need.

A second practical way to reduce the effects of drift is to recover the species as quickly as possible. Loss of genetic variation increases with decreasing population size, but such loss also increases dramatically if populations remain small over time (Hartl 1988). Current efforts to increase populations, and the lack of such efforts, have substantial effects on the total genetic variation that will be retained by the species in the future.

Catastrophes

Catastrophes are rare, irregularly occurring events that produce extreme changes in demography and population dynamics. There are two types of catastrophes that threaten red-cockaded woodpecker populations: catastrophic winds (hurricanes, downbursts, and tornadoes) and outbreaks of southern pine beetles. The beetles kill cavity trees, but not birds—at least not directly. It is possible that loss of foraging habitat and cavity trees to beetles could alter survival and productivity of woodpeckers, but this has not been demonstrated. Outbreaks of sufficient size to constitute a catastrophe at the

population level will probably be restricted to small populations dependent on tree species other than longleaf pine. Longleaf is sufficiently resistant to beetles to preclude outbreaks large enough to constitute a catastrophe. In other habitat types, the only real threat to population viability is loss of cavity trees, and this can be countered by construction of artificial replacement cavities. Appropriate forest management can minimize the likelihood of catastrophic outbreaks. Thus beetle outbreaks are not a serious concern in relation to population viability.

Hurricanes, however, are. The devastation wrought by Hurricane Hugo on the population inhabiting the Francis Marion National Forest demonstrated all too clearly that such storms can produce catastrophic changes in mortality (Hooper *et al.* 1990). Further, by eliminating all cavity trees on many territories Hugo resulted in a catastrophic increase in the rate of territory abandonment, beyond that attributable to mortality alone. Because of the distribution of red-cockaded woodpeckers, most populations face a significant risk from hurricanes, although there is little risk to some inland populations (Hooper and McAdie 1995). That hurricanes will regularly strike woodpecker populations is inevitable, and therefore any strategy to ensure species and population viability must address this form of catastrophe specifically.

The first element in addressing the hurricane threat is to reduce risk to the species as a whole by maintaining a number of populations that are broadly spaced geographically, and including as many inland populations as possible among them (Hooper and McAdie 1995). The second element is to reduce risk of extinction of individual populations through rehabilitation following the catastrophes that occur. The Hugo experience demonstrates that it is possible, albeit at considerable expense, to reduce impacts at the population level and facilitate recovery to pre-storm levels through proper management immediately following the storm (Watson *et al.* 1995). The critical activity is to construct artificial cavities quickly, and distribute them so that as few territories as possible are completely lacking in cavity trees. This will maximize the number of territories that remain occupied, which is the most critical component of population dynamics. It is anticipated that one or two recovery populations, as well as a number of support populations, will be in the process of recovering from storms at any given time (Hooper and McAdie 1995). Some support populations may be lost to hurricanes, despite proper rehabilitation efforts, but recovery populations should not be.

The third and final element in addressing the hurricane threat is to manage individual populations at risk to reduce their vulnerability to wind damage. Hooper and McAdie (1995) offer a number of suggestions, such as reducing access of wind into stands and creating conditions for growth that favor development of greater wind resistance. More research in this area is needed before a detailed policy can be developed, but managers of populations at risk should consider the factors discussed by Hooper and McAdie (1995) in developing their forest management plans.

A Strategy for Species Viability

The strategy to recover the red-cockaded woodpecker consists of recovering a number of individual populations, designated recovery populations, to levels at which they are individually viable against environmental stochasticity. Populations large enough to be resilient to environmental stochasticity will also be able to withstand threats from demographic stochasticity and inbreeding. Currently, our best estimate of the population size necessary to withstand effects of environmental stochasticity is 250 potential breeding groups. However, this is a minimum estimate based on model simulations, and it may contain some error. To be conservative, a number of larger populations (350 potential breeding groups) will exist at the time of recovery. These two population sizes, 250 and 350 potential breeding groups, are probably insufficient to avoid loss of genetic variation through genetic drift, at least in the absence of immigration. (Some researchers consider 350 breeding groups as the minimum size necessary to produce enough novel variation to offset loss from drift).

There are several strategies to reduce the loss of genetic variation as much as possible. First, recovery populations should be increased as far beyond the above population sizes as the habitat base will allow. Second, populations should be recovered as rapidly as possible, because loss of genetic variation increases with the length of time that populations remain small. Third, recovery populations represent the full range of habitat types now occupied by red-cockaded woodpeckers, and this range will aid the conservation of local genetic resources. Finally, dispersal between populations should be facilitated to the fullest extent possible. We have increased the total number of designated recovery populations in part to enhance the likelihood of natural dispersal among these populations once the species is recovered. We stress the importance of support populations as sources of immigrants to replace lost variability, and that support populations should be maintained until and after recovery. We recognize that translocation may need to be employed to maintain genetic variation within populations and species-wide, if natural dispersal is found to be insufficient.

Support populations should include 40 to 100 potential breeding groups, depending on spatial configuration, in order to eliminate demographic stochasticity and inbreeding depression as threats to their existence. If they can be maintained at even higher levels, their likelihood of extirpation due to environmental stochasticity will be reduced. Support populations that cannot meet the 40 to 100 size criterion can still serve the purpose of providing genetic variability to other populations, but extirpation of some of these is anticipated. We recommend that they be maintained at the largest size the habitat base will support.

The value of support populations depends on their genetic and spatial relationship to recovery populations. Value cannot be assessed precisely until more information about actual immigration, or how probability of immigration depends on distance and intervening habitat type, is available. The number of support populations required for each recovery population cannot be determined until information on levels of gene flow necessary to compensate for lost genetic variability is available. In the meantime, all

support populations, including those of less than 40 potential breeding groups, should be considered necessary to species viability.

The designated recovery populations were selected to eliminate the risk of extinction to the species as a whole due to hurricanes. Measures designed to reduce vulnerability to wind damage and to rehabilitate populations following storms should be sufficient to prevent extirpation of those individual recovery populations at risk. However, some support populations may be lost to hurricanes, with risk being a function of population size, location, and expected frequency of storms.

Populations must be managed to achieve favorable spatial configuration, as well as large size. Specifically, groups should be clustered to the extent possible, so that each group has multiple other groups within 3.2 km (2 mi). Special attention should be paid to the edges of the population, to keep isolation of individual groups there to a minimum.

In summary, the strategy to achieve species viability is to maintain a number of recovery populations that, with immigration from support populations, are individually viable to genetic and demographic threats. The threat to species viability from hurricanes is substantially reduced by maintaining a sufficient number of recovery populations, including inland ones, so that anticipated, periodic catastrophic reductions in some recovery populations do not threaten the species as a whole.

D. CAVITY TREE AND CLUSTER ECOLOGY

Red-cockaded woodpeckers are unique among North American woodpeckers in that they nest and roost in cavities they excavate in living pines (Steirly 1957, Short 1982, Ligon *et al.* 1986). This unusual behavior is thought to have evolved in response to the scarcity of snags and hardwoods in the fire-maintained pine ecosystems of the southeast (Ligon 1970, Jackson *et al.* 1986). Excavation of cavities in live pines has given rise to additional unusual and complex behaviors, ranging from cooperative breeding (Walters *et al.* 1992a; see 2B) to daily excavation of resin wells to create resin barriers against predatory rat snakes (*Elaphe obsoleta*, Ligon 1970, Dennis 1971b, Jackson 1974, 1978a, Rudolph *et al.* 1990b). Use of live pines is also the basis of the species' requirement for mature pine forests and woodlands, the loss of which has resulted in endangerment. Cavities are an essential resource for red-cockaded woodpeckers during the breeding season and also throughout the year, because they are used for both nesting and roosting. Thus, a thorough understanding of cavity tree ecology is fundamental to red-cockaded woodpecker biology, management, and recovery. This section describes current knowledge in support of the guidelines for management of cavity trees and clusters presented in 8D.

Cavity Excavation and Selection of Cavity Trees

Excavation of cavities in live pines is an amazingly difficult task. Birds must first select an old pine (Jackson and Jackson 1986, Conner and O'Halloran 1987, DeLotelle

and Epting 1988, Rudolph and Conner 1991), then excavate through 10 to 15 cm (4 to 6 in) of live sapwood, avoid dangerous pine resin that seeps from the cavity during excavation, and construct a cavity completely contained within the heartwood (Jackson 1977, Hooper *et al.* 1980, Conner and Locke 1982, Conner and O'Halloran 1987, Hooper 1988, Hooper *et al.* 1991b). Cavity excavation typically takes many years (Jackson *et al.* 1979, Rudolph and Conner 1991, Conner and Rudolph 1995a, Harding 1997).

The difficulty of cavity excavation is considered a major factor in the evolution of cooperative breeding in red cockaded woodpeckers (Walters 1990, Walters *et al.* 1988a, 1992a, 1992b; see 2B). Birds cannot easily exploit previously unoccupied habitat and build cavities, and so competition for territories with existing cavities is unusually intense. Young males delay reproduction and remain on their natal territory as helpers to increase their likelihood of obtaining a breeding site with existing cavities (Walters 1990, Walters *et al.* 1988a, 1992b). Natural formation of groups in previously unoccupied habitat (pioneering, Hooper 1983) is rare; its estimated annual rate is less than 3 percent of total groups in a population under current conditions (J. Walters, unpublished; see 2B).

Red-cockaded woodpeckers use a variety of pine species as cavity trees including longleaf, loblolly, shortleaf, slash, pond, pitch (*P. rigida*), and Virginia pines (*P. virginiana*; Steirly 1957, Lowery 1960, Mengel 1965, Sutton 1967, Hopkins and Lynn 1971, Jackson 1971, Murphy 1982). Longleaf, loblolly, and shortleaf pines are the most common species used for cavity trees and longleaf is considered preferred (Lowery 1960, Hopkins and Lynn 1971, Jackson 1971, Baker 1981, Bowman *et al.* 1997). All cavities are excavated in live pines, but occasionally woodpeckers roost and even nest in cavities in trees that have recently died (Hooper 1982, Patterson and Robertson 1983, R. Costa, pers. comm.).

Red-cockaded woodpeckers excavate cavities in live pines because they can exploit the pine's resin to protect against predation of nests and adults by arboreal snakes (Ligon 1970, Dennis 1971b, Jackson 1974, 1978a, Rudolph *et al.* 1990b). The birds create and maintain resin wells, or wounds in the cambium, to coat the trunk with resin which then effectively interferes with the snakes' ability to climb the tree (Rudolph *et al.* 1990b).

Longleaf pine is preferred for use as cavity trees because it produces more resin and can sustain resin flow for more years than other southern pines (Wahlenburg 1946, Hodges *et al.* 1977, 1979). The production of more resin affords the woodpeckers greater protection against snakes, and also provides the tree with greater protection against insects such as southern pine beetles (Hodges *et al.* 1979). Annual survival of longleaf cavity trees was twice that for loblolly and shortleaf cavity trees in east Texas, in part because of longleaf pine's greater resistance to southern pine beetles (Conner and Rudolph 1995a). Because of higher survival and the ability to sustain resin flow over time, longleaf pines may remain in use as cavity trees for several decades—much longer than shortleaf or loblolly pines (Conner and Rudolph 1995a, Harding 1997).

Cavity excavation time may be longer in longleaf pines than in either loblolly or shortleaf pines. In Texas, excavation time averaged 6.3 years in longleaf pines, two to three times greater than the average times for loblolly and shortleaf pines (Conner and Rudolph 1995a). In North Carolina, excavation times for cavities in longleaf averaged from 10 to 13 years, and from 6 to 9 years for loblolly (Harding 1997). Cavity excavation is an intermittent process, with month-long or longer breaks to allow resin flow to subside through resinosis (saturation of sapwood with hardened resin; Conner and Rudolph 1995a). Thus, longleaf may require longer excavation times because of its greater resin flow (Conner and Rudolph 1995a). Variation in estimated excavation times may result from geographic variation in resin flow (Harding 1997), itself a function of site and tree factors (Hodges *et al.* 1979, Ross *et al.* 1995), or from variation in research methods.

Selection of and Requirement for Old Trees

Red-cockaded woodpeckers select and require old pines for cavity excavation (Jackson and Jackson 1986, Conner and O'Halloran 1987, DeLotelle and Epting 1988, Rudolph and Conner 1991). Age of cavity trees depends on the ages of pines available, but there is a minimum age, generally 60 to 80 years depending on tree and site factors, below which use as a cavity tree is highly unlikely or simply not possible (DeLotelle and Epting 1988, Hooper 1988, Rudolph and Conner 1991). Currently, cavity trees average approximately 80 to 150 years in age and can be much older (Rudolph and Conner 1991, Hedrick 1992). Cavity trees are generally the oldest trees available in today's forests (Jackson *et al.* 1979, Engstrom and Evans 1990, Rudolph and Conner 1991), and the optimal age for cavity trees may be well above the average age of cavity trees under current forest conditions. For example, red-cockaded woodpeckers in national forests of Texas continue to select the oldest trees available for initiation of cavities, even though the forests have aged 20 years during the course of study (Rudolph and Conner 1991).

One reason red-cockaded woodpeckers require old trees for cavity excavation is that they need sufficient heartwood diameter at preferred cavity heights to construct the cavity completely within the heartwood. Cavities must be completely within the heartwood to prevent dangerous pine resin in the sapwood from entering the chamber (Jackson and Jackson 1986, Clark 1993), and the estimated minimum amount of heartwood required is 14.0 to 15.2 cm (5.5 to 6 in; Conner *et al.* 1994). Preferred cavity heights generally range from 6.1 to 15.2 m (20 to 50 ft; Baker 1971b, Hopkins and Lynn 1971, Hooper *et al.* 1980, Conner and O'Halloran 1987), a possible adaptation to minimize likelihood of ignition by frequent fire (Conner and O'Halloran 1987, Clark 1992, Conner *et al.* 1994). The age of the tree determines heartwood diameter at cavity height, as older pines have more heartwood at greater heights. In eastern Texas, longleaf pines between 70 and 90 years old had adequate heartwood at appropriate heights to contain a cavity (Conner *et al.* 1994). Fifty year-old longleaf pines examined by Clark (1992) had insufficient heartwood for cavity excavation.

A second reason that woodpeckers select old trees for cavity excavation is that old pines have a higher frequency of infection by red heart fungus, and the associated decay

of the heartwood becomes more advanced as the tree ages (Wahlenberg 1946). Woodpeckers can and do excavate cavities into undecayed heartwood (Beckett 1971, Conner and Locke 1982, Hooper 1988, Hooper *et al.* 1991b), but the presence of red heart fungus can substantially reduce the time required for cavity excavation (Conner and Rudolph 1995a). In Texas, for example, average excavation times for cavities in pines with and without decayed heartwood were 3.7 and 5 years, respectively (Conner and Rudolph 1995a).

Heartwood decay by red heart fungus was not frequently found in longleaf cavity trees in Texas until they were over 120 years old (Conner *et al.* 1994). Red heart is a very slow growing fungus (Affeltranger 1971, Conner and Locke 1982, 1983), and at least 12 to 20 years may be required between initial inoculation and the decay of sufficient heartwood to house a cavity (Conner and Locke 1983). Also, red heart fungus enters the heartwood of the tree through heartwood in large branches, and so trees must be old enough to have large branches before bole heartwood can be infected (Affeltranger 1971, Conner and Locke 1982). However, regional differences may exist in the ages and rates at which pines become infected with heartwood decaying fungi. A study in Texas reported a 46 percent infection rate for 50 longleaf cavity trees that averaged 126 years in age (Conner *et al.* 1994), whereas this rate was more than doubled for similarly aged longleaf cavity trees in South Carolina (97 percent infection rate for trees averaging 130 years in age; Hooper 1988).

Red-cockaded woodpeckers actively select pines with heartwood decayed by red heart fungus (Steirly 1957, Jackson 1977, Conner and Locke 1982, Hooper 1988, Hooper *et al.* 1991b, Rudolph *et al.* 1995). In fact, red-cockaded woodpeckers are able to detect and locate cavities in the specific area of the bole that is infected (Rudolph *et al.* 1995). Preference for decayed heartwood results in the selection of cavity trees that are older than necessary for them to have enough heartwood to contain a cavity (Hooper 1988, Hooper *et al.* 1991b, Rudolph *et al.* 1995). For example, cavity trees in Texas averaged 24.8 cm (9.75 in) in heartwood diameter, considerably larger than the 15.2 cm (6 in) estimated minimum (Rudolph *et al.* 1995). In fact, preference for red heart infection rather than age itself may drive the general preference for old trees (Hooper 1988).

Red-cockaded woodpeckers have been shown to select pines that have thinner sapwood and greater heartwood diameters than pines generally available nearby (Conner *et al.* 1994). This is also related to age: such trees are older, growing more slowly, and usually have a higher rate of red heart infection than pines not used for cavity excavation. Diameter growth of trees typically accelerates annually as younger trees mature, attains a maximum, and slows as trees approach maturity (Kramer and Kozłowski 1979). Heartwood diameter increases significantly with tree size and age in both loblolly and longleaf pines (Clark 1992, 1993).

Old-growth pines are relatively rare throughout the south. Old-growth remnants (both single trees and stands) within today's forests are critically important habitat and will continue to be so over the next 20 to 30 years, until second and third-growth forests mature and potential cavity trees become more widely available. Woodpeckers require

potential cavity trees in abundance throughout the landscape, because of currently high mortality of natural cavity trees and high rates of damage to existing cavities by pileated woodpeckers (*Dryocopus pileatus*; Conner *et al.* 1991a, Conner and Rudolph 1995b, Saenz *et al.* 1998; see 2F).

Selection of Trees with High Resin Production

Red-cockaded woodpeckers are known to select, as cavity trees, pines that have higher resin flow than surrounding pines (Bowman and Huh 1995, Conner *et al.* 1998a). Moreover, breeding males select the cavity tree with the highest resin flow for use as the nest tree (Conner *et al.* 1998a). Thus, woodpeckers benefit from pines with high resin production potential, likely indicated by high crown volume and crown weight (Conner and O'Halloran 1987). Ross *et al.* (1997) showed that longleaf pine cavity trees in low stand densities and on forest edges produced significantly more resin than similar cavity trees within interior forest stands with higher stem densities. Several studies have observed the tendency of red-cockaded woodpeckers to place their cavities near forest edges and in areas of low tree densities (Conner and O'Halloran 1987, Conner *et al.* 1991b, Ross *et al.* 1997), presumably because of higher resin flow in these locations.

The Cavity Tree Cluster

Each red-cockaded woodpecker in a group roosts in a cavity year-round, and it is usually the breeding male's cavity that holds the group's nest in the spring. The aggregation of active (in use) and inactive (previously used) cavity trees within an area defended by a single group is called the cluster (Walters *et al.* 1988a). Boundaries of clusters change as new cavity trees are excavated, old cavity trees fall, and existing cavity trees are captured or lost to neighboring woodpecker groups. The boundary of a cluster is generally considered to be the minimum convex polygon necessary to enclose all active and inactive cavity trees 'belonging to' a group of woodpeckers. For management purposes, the minimum cluster area size is 4.05 ha (10 ac), as some clusters may only contain one cavity tree. In order to protect cavity trees, some agencies have established a 61 m (200 ft) buffer zone of protection around the entire cluster. To facilitate record keeping and protection, individual cavity trees within a cluster are generally marked with metal numbered tags, painted for easy detection, and mapped.

Disturbance within the Cluster

Human-caused disturbances in cluster areas during the nesting season may disrupt red-cockaded woodpecker nesting activities, decrease feeding and brooding rates, and cause nest abandonment. Such activities may include but are not limited to all-terrain and other off-road vehicles, motorized logging equipment, and other vehicles that make excessive noise and disturbance to which the woodpecker groups have not previously become accustomed. Use of vehicles and other activities throughout the year may cause indirect impacts to red-cockaded woodpeckers through excessive soil compaction, damage to cavity tree roots, and disturbance of the groundcover. Soil compaction and root damage elevate cavity tree mortality (Nebeker and Hodges 1985, Hicks *et al.* 1987,

Conner *et al.* 1991a); changes in the groundcover may affect prey abundance (Collins 1998), nutrient value of prey (James *et al.* 1997), and fire frequency and intensity through changes in fuel.

Geographic Variation in Habitat

There is geographic variation in nesting and roosting habitat of red-cockaded woodpeckers. The largest populations tend to occur in the primarily longleaf pine forests and woodlands of the Coastal Plains and Carolina Sandhills (Carter 1971, Hooper *et al.* 1982, James 1995, Engstrom *et al.* 1996). Woodpeckers are also found in shortleaf/loblolly forests of the Piedmont, Cumberland, and Ouachita Mountain regions (Mengel 1965, Sutton 1967, Steirly 1973). Pine habitat occupied by red-cockaded woodpeckers covers a wide moisture gradient ranging from hydric slash pine (*P. ellioti* var. *densa*) flatwoods in Florida (Beever and Dryden 1992, Bowman and Huh 1995) to dry ridge and mountain tops in Oklahoma (Masters *et al.* 1989, Kelly *et al.* 1993), Kentucky (Kalisz and Boettcher 1991), Alabama, and Mississippi. Density of pine overstory in areas occupied by red-cockaded woodpeckers varies from fairly dense in Texas (Conner and O'Halloran 1987, Conner and Rudolph 1989), to sparse in the Orlando, Florida vicinity (DeLotelle *et al.* 1987), to extremely low in the Big Cypress National Preserve (Patterson and Robertson 1981).

Structure of Vegetation within Clusters

Alteration of the natural fire regime during the past century has caused fundamental changes in the vegetation structure of upland habitats throughout the south. These changes include a gradual encroachment of hardwoods, increasing dominance of off-site pine species such as slash and loblolly, and more densely wooded forests in general (Jackson *et al.* 1986, Ware *et al.* 1993). Each of these changes is detrimental to red-cockaded woodpeckers, and hardwood encroachment especially is a major cause of the species' decline and endangered status (see 1A).

The association of red-cockaded woodpeckers with open, park-like pine woodlands has long been known (Thompson and Baker 1971, Van Balen and Doerr 1978, Locke *et al.* 1983, USFWS 1985). Encroachment of hardwood midstory causes abandonment of cavity trees and clusters (Beckett 1971, Hopkins and Lynn 1971, Van Balen and Doerr 1978, Locke *et al.* 1983, Hovis and Labisky 1985, Conner and Rudolph 1989, Loeb *et al.* 1992). Cluster abandonment has been documented when hardwood and pine midstory basal area exceeds 5.7 sq. m per ha (25 sq. ft per acre; Conner and Rudolph 1989, Loeb *et al.* 1992). Negative effects of midstory growth above 3.7 m (12 ft) have also been shown (Hooper *et al.* 1980).

Thus, effective midstory control is an absolute prerequisite to management, conservation, and recovery of red-cockaded woodpeckers throughout their range. Such control is not an easy task. After seven decades of fire suppression, many clusters have developed an extensive hardwood component with an impressive underground root stock, particularly in the more mesic sites where loblolly and shortleaf pines are the dominant

tree species (Conner and Rudolph 1989). Repeated prescribed burning during the growing season can be an effective means to remove hardwoods and restore native groundcovers, and has the least detrimental impacts on soil structure and desired groundcovers (see 3G). However, excessive quantities of hardwoods (or very large trees) may require removal by hand, mechanical means (Conner *et al.* 1995), one-time herbicide application (Conner 1989), or a combination of these methods prior to restoration burning. Maintenance of open habitat structure is best achieved through use of growing-season fire fueled by native grasses.

Removal of hardwood midstory and thinning of overstory pines in clusters outside of the nesting season does not negatively affect red-cockaded woodpeckers (Conner and Rudolph 1991a), but mechanical removal of midstory should not be done when red-cockaded woodpeckers are nesting (Jackson 1990). If clusters have been abandoned due to unsuitable habitat conditions, they should be conserved and restored to suitable midstory conditions to increase the probability of reoccupation by woodpeckers (Doerr *et al.* 1989).

Red-cockaded woodpeckers can tolerate some hardwood overstory trees (basal area less than 2.3 sq. m per ha; 10 sq. ft per acre) within clusters (Hooper *et al.* 1980, Hovis and Labisky 1985, Conner and O'Halloran 1987). Small numbers of overstory hardwoods or large midstory hardwoods at low densities are consistent with historic landscapes in some habitats, and do not have the same negative effects on red-cockaded woodpeckers as the dense hardwood midstories resulting from fire suppression. Oak inclusions and upland hardwood species, such as post oak (*Quercus stellata*) and bluejack oak (*Q. incana*), occur naturally in association with the pine ecosystems of the south. Such species are integral components of the southern pine ecosystem and should not be cut in the name of red-cockaded woodpecker management.

Stream drainages, with associated shrub and midstory layers and hardwoods, are also integral parts of the southern pine ecosystems. However, woodpeckers may not be able to tolerate the complex vegetative structure of stream drainages near cavity trees. Therefore, management of cavity tree habitat for red-cockaded woodpeckers should be primarily focused in upland portions of the forest landscape. Stands developed and managed to recruit new woodpecker groups or replace cluster habitat should be located away from stream drainages whenever possible.

Density of pines in clusters varies according to habitat type, geography, and silvicultural history. The sparsest woods occupied by red-cockaded woodpeckers are the hydric slash pine woodlands of south Florida (Beever and Dryden 1992). Slightly more dense are the clusters in longleaf woodlands of south and central Florida; average basal area of clusters in these Florida longleaf woodlands currently ranges from 1.8 to 5.7 sq. m per ha (8 to 25 sq. ft per acre; DeLotelle *et al.* 1983, Shapiro 1983, Hovis and Labisky 1985, Bowman *et al.* 1997). For clusters in longleaf pine woodlands north of Florida, estimated average basal area ranges from 9.2 to 13.8 sq. m per ha (40 to 60 sq. ft per acre) of basal area (Crosby 1971, Hopkins and Lynn 1971, Thompson and Baker 1971). Clusters in natural loblolly and/or shortleaf pine forests average slightly higher densities

(Thompson and Baker 1971, Hooper *et al.* 1980, Conner and O'Halloran 1987, Conner and Rudolph 1989).

Woodpecker cluster stands are typically less dense than surrounding stands (Crosby 1971, Thompson and Baker 1971, Grimes 1977, Locke *et al.* 1983, Shapiro 1983, Wood 1983, Hovis and Labisky 1985, Conner and O'Halloran 1987, Conner *et al.* 1991b, Loeb *et al.* 1992, Bowman *et al.* 1997) and they may be the least dense stands available. For example, Conner *et al.* (1991b) reported a preference for seed-tree and shelterwood cuts for cavity excavation in longleaf pine woodlands. For clusters, basal areas as low as 9.2 sq. m per ha (40 sq. ft per acre) in longleaf stands and from 9.2 to 13.8 sq. m per ha (40 to 60 sq. ft per acre) in shortleaf/loblolly stands are better than higher basal areas, and even lower basal areas may be suitable (Conner *et al.* 1991b). However, seed-tree and shelterwood cuts with excessive pine or hardwood midstory are not acceptable as nesting habitat.

Low density of cluster pines confers many benefits to red-cockaded woodpeckers. Pines in low density stands grow larger in diameter, have greater crowns and root systems, and higher resin flow. Such pines are more resistant to wind damage and attacks by bark beetles, may be used as cavity trees at younger ages, and provide woodpeckers with greater protection against predation. In addition, sparse woods may have a greater proportion of area in grass and forb groundcovers than more dense forests, and these groundcovers in turn affect arthropod abundance (Collins 1998) and the ability of the stand to carry fire. Another reason for the preference for sparsely wooded stands, apart from the above benefits, may be that the low density of pine itself is a reflection of frequent fire.

Cavity Tree Mortality and Nesting Habitat Protection

Infestation by southern pine beetles is the major cause of cavity tree mortality in loblolly and shortleaf pines (Conner *et al.* 1991a). For southern pines, defense against bark beetle attack is positively related to the trees' ability to produce oleoresins (Lorio 1986). Cavity trees may be lost to southern pine beetles during epidemics, such as the death of 350 cavity trees including more than 50 entire clusters during the early 1980's in the Sam Houston National Forest (Conner *et al.* 1991a, 1997a). Cavity trees are also lost to southern pine beetles at endemic population levels, at a lower but steady rate (Conner *et al.* 1997a). Thus, cavity tree mortality due to both epidemic and endemic southern pine beetles can substantially impact woodpeckers, particularly small populations in the loblolly and shortleaf pines of Texas, Arkansas, Louisiana, Mississippi, and elsewhere (Conner and Rudolph 1995b, Rudolph and Conner 1995). Factors that increase loss of cavity trees and other important (mature) pines in the cluster include physical disturbance of soils and roots during thinning and midstory removal, high density of pines within the cluster, and excessive hardwood midstory outside the cluster (Thatcher *et al.* 1980, Nebeker and Hodges 1985, Hicks *et al.* 1987, Conner *et al.* 1997a). Excessive midstory outside the cluster, a common characteristic of lands currently managed for red-cockaded woodpeckers, can funnel the insects toward the cluster (Conner *et al.* 1997a). Fortunately, pines with artificial cavities, used to mitigate losses of cavity trees to

southern pine beetles, are not infested at a rate significantly different from pines with naturally excavated cavities (Conner *et al.* 1998b). Managers should favor pines with high resin producing ability. Managers should also reduce risk of beetle infestation through pine thinning and minimizing disturbance during periods of high beetle activity (Mitchell *et al.* 1991). Stands where loblolly and shortleaf pines are the dominant cover type need to be thinned regularly to maintain basal areas less than 18.4 sq. m per ha (80 sq. ft per acre) or to keep an average spacing of at least 7.6 m (25 ft) between pines in mature stands, to retard the spread of beetle infestations (Thatcher *et al.* 1980, Hicks *et al.* 1987, Nebeker and Hodges 1985, Mitchell *et al.* 1991).

Wind is the second greatest cause of cavity tree mortality in non-hurricane situations (Conner *et al.* 1991a). Cavity trees can be uprooted or snapped by high velocity winds. Patterns of harvest near clusters should be carefully planned to avoid funneling wind toward cavity trees (Conner *et al.* 1991a, Conner and Rudolph 1995c). A forest buffer of uncut trees greater than 61 m (200 ft) around cavity trees is adequate protection to minimize wind damage, wind snap, and wind throw during isolated severe summer thunderstorms (Conner and Rudolph 1995c).

Hurricane winds are a major threat to coastal woodpecker populations (Engstrom and Evans 1990, Hooper *et al.* 1990, Hooper and McArdie 1995, Lipscomb and Williams 1995). For example, when Hurricane Hugo struck the Francis Marion National Forests in South Carolina during September 1989, it destroyed 87 percent of the cavity trees, 67 percent of the woodpeckers, and 70 percent of the foraging habitat (Hooper *et al.* 1990, Hooper and McArdie 1995). Drilled and inserted artificial cavities (Copeyon 1990, Allen 1991, Taylor and Hooper 1991), having just been developed, enabled the rapid recovery of the Francis Marion population (Watson *et al.* 1995). Conservation of inland populations and many separate coastal populations will minimize the risk of extinction from hurricanes (USFWS 1985, Hooper and McArdie 1995). Hooper and McArdie (1995) also suggest that pines needed for future nesting habitat be grown in open conditions to promote the development of large crowns, extensive root systems, and strong boles. Another strategy to minimize impacts from hurricane winds is to avoid the creation of openings greater than 10.1 ha (25 ac) in or near forests managed for red-cockaded woodpeckers in hurricane-prone areas.

The third major cause of cavity tree mortality is fire. Managers must take appropriate measures to protect cavity trees from prescribed burns and wildfires so that loss is minimized. Foremost among these protective measures is regular burning within the cluster and around cavity trees, to keep fuel at acceptable levels. Other techniques are described in 8I.

Implications for Management

Cavities, cavity trees, and cavity tree clusters currently limit red-cockaded woodpecker populations, and thus their careful management is foremost in woodpecker conservation and recovery. Red-cockaded woodpeckers require large old trees as nesting and roosting sites, in habitat that is free of pine and hardwood midstory. Each cavity tree

is an important resource that must be protected, and until potential cavity trees become more widely available, additional cavities and clusters must be provided through the use of artificial cavity technology. Hardwood encroachment causes abandonment of cavity tree clusters, with direct effects on population status. Invasive hardwoods must therefore be controlled, preferably by growing season fire. These management actions—protection of existing cavity trees, provisioning of artificial cavities and clusters as appropriate, and hardwood control—form the basis of red-cockaded woodpecker management (see 8A-H for more information). Loss of cavity trees and hardwood encroachment were primary factors in the decline of the species throughout its range (see 1A). Removal of these limiting factors is therefore fundamental to recovery.

E. FORAGING ECOLOGY

Our understanding of the foraging ecology of red-cockaded woodpeckers is increasing, although much work remains to be done. Natural geographic variation in forest ecology and woodpecker demography as well as the highly altered structure of today's forests make documenting habitat preferences and requirements a complex and challenging task. Despite these difficulties, a body of research has been developed describing foraging ecology and habitat relationships of red-cockaded woodpeckers. Here, we summarize research into diet, habitat selection, and habitat effects on fitness. In 8G, we present guidelines for providing foraging habitat that is suitable in quality and quantity based on current knowledge. Further research will help us to better understand foraging habitat requirements and may result in revisions of present guidelines.

Diet and Prey Abundance

Diet of Adults and Nestlings

Over 75 percent of the diet of red-cockaded woodpeckers consists of arthropods, especially ants and roaches, but also beetles, spiders, centipedes, true bugs, crickets, and moths (Beal *et al.* 1941, Baker 1971a, Harlow and Lennartz 1977, Hanula and Franzreb 1995, Hess and James 1998). Ants are particularly common in the diet of adults, comprising over half the stomach contents of adults and sub-adults in the Gulf coast region (Beal *et al.* 1941) and the Apalachicola National Forest in Florida (Hess and James 1998). Other arthropods comprised an estimated 34 percent and 17 percent, respectively, of the adult diet in these two studies (Beal *et al.* 1941, Hess and James 1998). *Crematogaster ashmeadii* may be the most prominent of the ant species in the woodpecker diet, especially on the Gulf Coast (Beal *et al.* 1941, Hess and James 1998). *C. ashmeadii* comprised 74 percent of the ant biomass taken by birds in Florida, whereas *Solenopsis picta*, *Camponotus nearticus*, and *C. floridanus* constituted 19 percent, 7 percent, and 1 percent of ant biomass respectively (Hess and James 1998).

Fruits and seeds make up the small remaining portion of the adult diet. Red-cockaded woodpeckers have been known to eat the fruits or seeds of pines (*Pinus spp.*), poison ivy (*Rhus radicans*), magnolia (*Magnolia spp.*), myrtle (*Myrica spp.*), wild cherry

(*Prunus serotina*), wild grape (*Vitus spp.*), blueberry (*Vaccinium spp.*), and blackgum (*Nyssa sylvatica*). Fruits and seeds comprised 14 percent by volume of the stomach contents of adults collected throughout the year in the Gulf Coastal Plain (Beal *et al.* 1941). Similarly, fruits and seeds made up 16 percent of the yearly diet of adults and 23 percent of the diet of non-breeding adults in Florida (Hess and James 1998). In contrast, plant material was rarely seen in the diets of woodpeckers in the Francis Marion National Forest of South Carolina (Harlow and Lennartz 1977, Hooper and Lennartz 1981).

The diet of nestlings differs from that of adults, and may also differ geographically. Nestlings are typically fed arthropods almost exclusively, although fruits may be given on occasion (Baker 1971a, Harlow and Lennartz 1977, Hanula *et al.* 2000b, J. Hanula and T. Engstrom, unpublished). Larger prey items are commonly fed to nestlings in addition to or instead of ants (Hanula and Franzreb 1995, Hess and James 1998, Hanula *et al.* 2000b), and there is some evidence that breeding groups increase their reproductive success by feeding large prey (Schaefer 1996). In the Apalachicola National Forest, the diet of nestlings (as estimated by stomach contents) consisted mainly of roughly equal proportions of ants, beetles, spiders, and centipedes (Hess and James 1998). In several populations in Georgia and South Carolina, wood roaches were the most common item fed to nestlings, comprising from 26 to 62 percent of the nestling diet (as estimated from photographs of feeding visits; Hanula and Franzreb 1995, Hanula *et al.* 2000b, J. Hanula and T. Engstrom, unpublished).

Prey Selection, Location, and Abundance

Red-cockaded woodpeckers generally capture arthropods on and under the outer bark of live pines and in dead branches of live pines. Pines that have recently died are also a notable source of prey (Ligon 1968, Hooper and Lennartz 1981, Schaefer 1996, Bowman *et al.* 1997). Red-cockaded woodpeckers rarely excavate through the bark of live pines to capture prey, but do excavate into dead branches (Ligon 1968, Ramey 1980, Hooper and Lennartz 1981, Porter and Labisky 1986, Schaefer 1996).

Differences in foraging behavior between the sexes in red-cockaded woodpeckers are well-documented (Ligon 1970, Hooper and Lennartz 1981, Engstrom and Sanders 1997, Hardesty *et al.* 1997). Males commonly forage in the crown of the tree, and are often on dead branches. Females commonly forage on the trunk, especially the lower trunk, and rarely forage on dead branches. This difference may serve to expose males and females, separately, to the areas of the tree with highest concentrations of ants and other arthropods (Hooper 1996, Hanula and Franzreb 1998). Recently, C. Rudolph *et al.* (unpublished) suggested that foraging behaviors differ by social status as well as sex. Breeding males may spend more time in the inner crown of the tree, whereas helper males may forage more on the crown's outer branches (C. Rudolph *et al.*, unpublished).

Several studies have assessed abundance and location of potential prey of red-cockaded woodpeckers (Hooper 1996, Hanula and Franzreb 1998, Hess and James 1998, Hanula *et al.* 2000a). Relative abundance of arthropods changes depending on the part of the tree sampled. Ants appear to be by far the most common arthropod on dead branches

(Hooper 1996, Hanula and Franzreb 1998). On the boles of the tree, the most abundant arthropods were true bugs, spiders, and roaches (Hooper 1996). On live branches, roaches, spiders, beetles and ants were most common (Hooper 1996). A large proportion of the arthropods on pine trees have gotten there by crawling up from the ground, which points to the condition of the ground cover as an important factor influencing abundance of prey for red-cockaded woodpeckers (Hanula and Franzreb 1998).

Relative abundance of different ant species on pine trees has been assessed in South Carolina (Hooper 1996, Hanula and Franzreb 1998) and in Florida (Hess and James 1998). In coastal South Carolina, ants of the genus *Crematogaster* were equal in abundance to those of *Camponotus* (Hooper 1996). In contrast, *Crematogaster* was three times more common than *Camponotus* in the Savannah River Site (Hanula and Franzreb 1998), and five times more abundant in the Apalachicola National Forest than all other ant species combined (Hess and James 1998). *Solenopsis picta*, while somewhat common in Florida (Hess and James 1998), was rare in South Carolina (Hooper 1996, Hanula and Franzreb 1998).

Thus, several studies have documented the prominence of ants, especially *C. ashmeadii*, in the diet of adult red-cockaded woodpeckers, and others have described patterns of arthropod abundance and distribution. Whether birds are selecting ants in greater proportion than their availability remains unknown. Assessing prey selection is extremely difficult, in large part because of extraordinary variability in the distributions of arthropods but also because each method of studying diet has its bias. In addition, diets of both adults and nestlings are highly variable: ants, for example, comprised from 0 to 94 percent of the stomach contents of nestlings and from 4 to 95 percent of the stomach contents of adults in Florida (Hess and James 1998). Nor is it clear whether plant material is a preferred or sub-optimal food. Plants may be selected to fill a nutritional need or exploited when prey is scarce.

Factors Affecting Prey Abundance

Arthropod abundance and biomass increases with the age and size of pines (Hooper 1996, Hanula *et al.* 2000a). Hanula *et al.* (2000a) found that arthropod abundance per tree increases linearly with stand age (the oldest stands with adequate sample sizes in their study were 70 to 75 years in age), and that arthropod biomass per tree increases with stand age until at least age 60. Their study showed a similar, positive relationship between arthropods and tree diameter, and negative relationships between density of pines and arthropod abundance and biomass per tree. It is not yet clear which factors—size, age, and/or density—are more important in determining arthropod abundance and distribution. Further research is required before the relationships among tree age, size, and density and prey abundance are fully understood.

Fire frequency also affects both arthropod abundance and species composition (James *et al.* 1997, Collins 1998). In Texas, arthropod abundance was higher in loblolly/shortleaf stands with grass and forb groundcover than in stands with substantial hardwood midstory (Collins 1998). Hanula *et al.* (2000a) documented positive

relationships between tree age and the abundance of both herbaceous groundcovers and insects, although there were no direct relationships between measures of herb and insect abundance.

Most importantly, however, a positive relationship between fire frequency (as shown by groundcover) and fitness of red-cockaded woodpeckers has been documented in several recent studies (James *et al.* 1997, in press, Hardesty *et al.* 1997). James *et al.* (in press) specifically documented an increase in fledging rate following the reintroduction of growing season fire, relative to control plots burned during the dormant season.

Frequent fire increases fitness of red-cockaded woodpeckers through more than one mechanism: first, by reducing hardwoods, and secondly, by increasing abundance and perhaps nutrient value of prey (James *et al.* 1997, Provencher *et al.* 1998). The increase in insect abundance is at least partially independent of the reduction in hardwoods. James *et al.* (1997) revealed this independence by showing an effect of fire on fitness in a study area that had few hardwoods. Provencher *et al.* (1998, unpublished) documented two to seven-fold increases in insect densities following growing season fire of hardwood-encroached longleaf stands. They then showed that reductions in hardwoods by herbicides and mechanical felling did not result in similar increases in insect densities until the stands were burned during the growing season (L. Provencher, unpublished). Thus, frequent growing season fire may be critically important in providing red-cockaded woodpeckers with abundant prey.

Selection of Foraging Habitat

Throughout their range, red-cockaded woodpeckers use open pine habitats for foraging. Considerable geographic variation in habitat types exists, illustrating the species' ability to adapt to a wide range of ecological conditions within the constraints of mature or old-growth, open southern pine systems. Woodpeckers use such habitat types as longleaf pine savannahs, flatwoods, sandhills, and clayhills; slash pine savannahs and flatwoods; pond and/or slash pine pocosins; shortleaf pine savannahs and forests, and shortleaf/loblolly pine savannahs and forests (Nesbitt *et al.* 1978, Ramey 1980, DeLotelle *et al.* 1983, Hooper and Harlow 1986, Porter and Labisky 1986, Bradshaw 1995, Epting *et al.* 1995, Bowman *et al.* 1997). Longleaf pine habitats are considered the optimal habitat type for red-cockaded woodpeckers and may have constituted nearly half of all occupied habitat prior to European colonization of North America (Conner *et al.*, in press).

Red-cockaded woodpeckers show a strong preference for living pines as foraging substrate (Hooper and Lennartz 1981, Porter and Labisky 1986, Jones 1994, Bowman *et al.* 1997). Pines used for foraging include longleaf, slash, loblolly, shortleaf, Virginia, and pond. Sand pine may be used rarely (Hardesty *et al.* 1997), and cypress is used on occasion, averaging an estimated 10 percent of foraging time in south-central Florida (Nesbitt *et al.* 1978, DeLotelle *et al.* 1983). Hardwoods are also used on occasion. Use of hardwoods typically accounts for 0 to 5 percent of foraging time (Hooper and Lennartz

1981, Repasky 1984, Porter and Labisky 1986, Bradshaw 1995, Hardesty *et al.* 1997). Reports of somewhat higher use include 7 percent of foraging time in Louisiana (Jones 1994) and 12 percent in Kentucky (Zenitsky 1999). Thus, hardwoods comprise a trivial or minor component of foraging substrate for red-cockaded woodpeckers throughout their range.

Dying and recently dead pines are an important foraging resource for red-cockaded woodpeckers (Ligon 1968, Hooper and Lennartz 1981, Schaefer 1996, Bowman *et al.* 1997). Pines infested with southern pine beetles provide an especially important though unpredictable food source, particularly in shortleaf and loblolly habitats (Schaefer 1996, Rudolph *et al.*, in prep.). In fact, southern pine beetles may be a substantial factor supporting red-cockaded woodpeckers in such habitats, both historically and currently. Because bark beetle-infested pines are extremely valuable as food sources, removal of recently dead and dying trees in any habitat is likely to adversely impact red-cockaded woodpeckers when southern pine beetles are at endemic population levels. However, if beetle populations are high and pines near cavity trees (or cavity trees themselves) are infested, some pines are generally removed in the attempt to control growing beetle infestations and prevent loss of nesting and foraging habitat.

Selection of Tree Species

Whether red-cockaded woodpeckers prefer to forage on a particular species of pine has not been clearly demonstrated, and it may be that no such preference exists. Previous research has yielded conflicting results, all of which could be confounded by other factors such as tree age and size, density of surrounding trees, and presence of hardwood midstory. Longleaf pine stands were selected over slash pine stands in northern Florida (Porter and Labisky 1986), but elsewhere in Florida slash pines were selected over longleaf (Nesbitt *et al.* 1978). Bowman *et al.* (1997) suggested that slash pine in south central Florida may provide important foraging in addition to longleaf. In the North Carolina Sandhills, woodpeckers did not select trees based on tree species, but over 90 percent of available pines were longleaf (Walters *et al.* 2000). Woodpeckers in coastal North Carolina did not select among longleaf, loblolly, and pond pines, even though the proportion of loblolly and pond pines together averaged over 20 percent of available pines (Zwicker and Walters 1999). Finally, it may be that in habitats that were traditionally longleaf, dominance of longleaf was sufficient to retard the evolution of selection among pine species by red-cockaded woodpeckers. Future research in habitat containing mixed pine species both historically and currently would help document the presence or absence of this behavior.

Selection of Older and Larger Trees

All studies examining selection of individual trees by foraging red-cockaded woodpeckers have found that the birds select large, old trees over small, young trees (Hooper and Lennartz 1981, Porter and Labisky 1986, DeLotelle *et al.* 1987, Bradshaw 1995, Jones and Hunt 1996, Engstrom and Sanders 1997, Hardesty *et al.* 1997, Zwicker and Walters 1999, Walters *et al.* 2000). Reports vary as to the specific sizes at which

trees are avoided and preferred. Also, some researchers suggest that all trees over a specific size (generally, 25.4 cm [10 in] dbh) are equal in foraging value (Hooper and Harlow 1986), whereas others suggest that foraging value of trees increases continually with increasing size and age of trees (Engstrom and Sanders 1997, Hardesty *et al.* 1997, Doster and James 1998, Zwicker and Walters 1999, Walters *et al.* 2000). Such disagreements are likely due to differences in study methods and to differences in available habitat, because what the birds select or avoid must always be a subset of what is available. Available habitat changes because of natural geographic variation but also because of variation in the extent of forest alteration (e.g., fire suppression and tree cutting). Despite the disagreements, it is clear that tree age and size strongly influence selection of pines for foraging. Results of previous studies are summarized below.

Reported sizes below which trees are avoided (that is, used less than their availability) varies from 12.7 cm (5 in) dbh in coastal South Carolina (Hooper and Lennartz 1981) to 20.3 and 25.4 cm (8 and 10 in) dbh in northwest Florida (Porter and Labisky 1986, Hardesty *et al.* 1997) and Louisiana (Jones and Hunt 1996), and 25.4 cm (10 in) dbh in the North Carolina Coastal Plain and Sandhills (Zwicker and Walters 1999, Walters *et al.* 2000). Reported sizes above which trees are selected (used more than their availability) include 20.3 and 25.4 cm (8 and 10 in) dbh in northwestern Florida (Porter and Labisky 1986, Hardesty *et al.* 1997), 25.4 cm (10 in) dbh in coastal South and North Carolina (Hooper and Lennartz 1981, Zwicker and Walters 1999), 30.5 cm (12 in) dbh in southwestern Georgia (Engstrom and Sanders 1997), the North Carolina Sandhills (Walters *et al.* 2000), coastal Virginia (Bradshaw 1995), and Arkansas (Doster and James 1998), and 40 cm (15.7 in) in Louisiana (Jones and Hunt 1996). Again, these differences are due in part to differences in available habitat, because what the birds select or avoid depends on what is there.

Fewer studies have assessed specific ages at which individual pines are avoided or selected, although several more have assessed effects of average stand age (see below). Age and size of trees are highly correlated, and at present it is not known whether tree age, size, or both age and size is most important to foraging woodpeckers. In the Coastal Plain and Sandhills of North Carolina, trees under 60 years in age were avoided whereas those over 60 years (Coastal Plain) and 70 years (Sandhills) were selected (Zwicker and Walters 1999, Walters *et al.* 2000). In northwestern Florida, trees less than 50 years in age were avoided, trees 50 to 150 years in age were used in proportion to their availability, and trees 150 years in age and older were preferred (Hardesty *et al.* 1997).

A preference by woodpeckers for the oldest and largest trees available has been shown in several studies (Hardesty *et al.* 1997, Engstrom and Sanders 1997, Zwicker and Walters 1999, Walters *et al.* 2000). Bradshaw (1995) also reported a preference for the largest trees, although he combined all trees over 30.5 cm (12 in) dbh into one category. Such preference for the oldest and largest trees available suggests that tree selection by red-cockaded woodpeckers may be operating in either of two ways: (1) woodpeckers always select the oldest and largest trees in any habitat, or (2) an optimal size and age exists above which selection becomes equal, but this optimum remains unseen because currently these trees are not generally available in meaningful amounts (Zwicker and

Walters 1999). In contrast, other studies report that selection tapers off above middle-aged, medium-sized trees—suggesting that middle-aged trees are of equal importance to the oldest and largest trees (Hooper and Lennartz 1981, Hooper and Harlow 1986). Again, such disagreements are likely due to differences in study methods and available habitat. As public forests regain an old-growth component and research methods are standardized, biologists will likely reach a consensus on what ages and sizes of trees are preferred by foraging red-cockaded woodpeckers.

Patch Selection

Habitat selection at a scale larger than individual trees, but smaller than stands, is referred to here as patch selection. Patch selection by red-cockaded woodpeckers has been explored in three studies. Bowman *et al.* (1997) found that woodpeckers foraged in patches containing fewer but larger trees than patches chosen randomly. Walters *et al.* (2000) found that woodpeckers used patches containing larger trees and lower hardwood midstory than unused patches. Doster and James (1998) found that red-cockaded woodpeckers prefer to forage in patches containing larger pines, a lower overstory pine density, and less hardwood midstory than randomly chosen patches nearby.

Stand Selection

Use of stands by red-cockaded woodpeckers is influenced by the size of the stand, stand age, density of pines, density of large pines, fire history (hardwood midstory), season, and proximity to cavity trees and territorial boundaries (Hooper and Harlow 1986, Porter and Labisky 1986, DeLotelle *et al.* 1987, Epting *et al.* 1995, Bradshaw 1995, Walters *et al.* 2000). Two studies documented a positive relationship between stand use and stand age after controlling for effects of cavity trees and territorial boundaries (DeLotelle *et al.* 1987, Epting *et al.* 1995). Porter and Labisky (1986) reported that preferred stands were much older than avoided stands (mean stand age = 72 and 18 years, respectively). Similarly, Jones (1994) reported that stands of trees less than 50 years old were avoided, and stand use increased continually with increasing stand age (Jones 1994, Jones and Hunt 1996). Hooper and Harlow (1986) also reported a positive effect of stand age on use but considered it to be weak.

Stand use and density of all pines may be positively related if densities are generally low (DeLotelle *et al.* 1987) and unrelated or negatively related if densities are high (Hooper and Harlow 1986, Bradshaw 1995). Effects of pine density on stand use also changes depending on the size of trees in question: increasing density of large trees is beneficial (Hooper and Harlow 1986, Bradshaw 1995, Walters *et al.* 2000), whereas high densities of small pines is detrimental (Porter and Labisky 1986, Walters *et al.* 2000). For example, stand use increased with increasing density of pines greater than or equal to 30.5 cm (12 in) dbh in Virginia (Bradshaw 1995), 35.6 cm (14 in) dbh in central North Carolina (Walters *et al.* 2000), and 22.9, 35.6, and 48.3 cm (9, 14, and 19 in) dbh in coastal South Carolina (Hooper and Harlow 1986, although they considered these effects to be weak and, for the largest size class, due mainly to the presence of cavity trees.) Stand use decreased with increasing densities of pines less than 25.4 cm (10 in)

dbh in central North Carolina (Walters *et al.* 2000); similarly, dense stands of young trees (average 559 stems/acre and 18 years in age) were avoided in northwest Florida (Porter and Labisky 1986).

Hardwoods appear to have a negative influence on stand use. Stand use decreased with increasing density of hardwoods in several studies (Hooper and Harlow 1986, Epting *et al.* 1995, Bradshaw 1995, Jones and Hunt 1996), and stand use was negatively influenced by the average height of midstory hardwoods in North Carolina (Walter *et al.* 2000). Jones and Hunt (1996) found that stands in which greater than 10 percent of canopy trees were hardwoods were avoided.

Finally, during the non-breeding season red-cockaded woodpeckers may travel long distances to access open stands of large pines, whereas during the breeding season birds may use stands containing smaller pines or a greater hardwood component if they are near nest cavities (Bradshaw 1995, Jones and Hunt 1996).

Home Range and Habitat Quality

Size of home ranges of red-cockaded woodpeckers have been described over much of the species' range and in several habitat types (Hooper *et al.* 1982, Wood 1983, Nesbitt *et al.* 1983, Repasky 1984, Porter and Labisky 1986, DeLotelle *et al.* 1987, Epting *et al.* 1995, Bradshaw 1995, Engstrom and Sanders 1997, Bowman *et al.* 1997, Hardesty *et al.* 1997, Doster and James 1998, Walters *et al.* 2000). In studies with sample sizes of over 10 groups, average year-round home range size was estimated to be 83.0 ha (205 ac) in south-central North Carolina (Walters *et al.* 2000), 87.0 ha (215 ac) in coastal South Carolina (Hooper *et al.* 1982), roughly 80.1 ha (198 ac) in coastal Georgia (Epting *et al.* 1995), 129.0 ha (319 ac) in central Florida (DeLotelle *et al.* 1995), and 108.9 ha (269 ac) in northwestern Florida (Hardesty *et al.* 1997). In addition, notable studies among those estimating home range based on fewer than 10 groups include one study in the northern edge of the species' current range (Bradshaw 1995), one in the southern edge of the species current and historic range (Nesbitt *et al.* 1983), and one in extremely rare old-growth longleaf forest in southwest Georgia (Engstrom and Sanders 1997). Bradshaw (1995) reported that average year-round home range size for 6 groups in coastal Virginia was 120.2 ha (297 ac); Nesbitt *et al.* (1983) estimated that summer range for 5 groups in south Florida was 144.5 ha (357 ac); and Engstrom and Sanders (1997) reported that home range size for 7 groups in old-growth forest in southwest Georgia was 46.9 ha (116 ac), the smallest average size yet reported (based on all-day follows). Also, Doster and James (1998) reported an average home range of only 24.7 ha (61 ac) for 5 groups of woodpeckers in shortleaf pine habitat of Arkansas, but this estimate was not based on all-day follows because rough terrain inhibited data collection.

Thus, home ranges in Florida tend to be larger than those farther north (DeLotelle *et al.* 1987, Hardesty *et al.* 1997), and those in fire-maintained old-growth forest are substantially smaller than those in second-growth (Engstrom and Sanders 1997). Larger samples would be helpful in confirming these effects, but are not available in several cases (e.g., Virginia Coastal Plain, old-growth forest). Together these results suggest that

the natural size and density of pines as well as degree of forest alteration (such as history of harvests and fire suppression) impact home range size. The size of a home range or territory may also increase if it is not constrained by the presence of neighboring groups (DeLotelle *et al.* 1987).

Several studies have related variation in home range (or territory) size within a population to habitat characteristics of the home range (Hooper *et al.* 1982, Bowman *et al.* 1997, Hardesty *et al.* 1997, Walters *et al.* 2000). Hooper *et al.* (1982) reported that for 24 groups in coastal South Carolina, territory size generally increased with increasing pine density and basal area. In contrast, Hardesty *et al.* (1997) reported that for 25 groups in northwest Florida, home range size decreased with increasing pine density and basal area. Walters *et al.* (2000) found home range size of 30 groups in south-central North Carolina was independent of pine density and basal area, but increased with increasing invasion by hardwoods. Thus, home range size depends on the quality of available foraging habitat: less habitat is needed if the quality of that habitat is high. Increasing pine density may be beneficial if pine density is low, or detrimental if density is high. In summary, studies of home range size suggest that red-cockaded woodpeckers require from 40.5 to 161.9 ha (100 to 400 ac) per group, depending upon the quality of foraging habitat, and that high quality foraging habitat is intermediate in pine density and relatively free of hardwood midstory. These characteristics of high-quality foraging habitat are consistent with those suggested by analyses of patch and stand selection (above) and group fitness (below).

Group Fitness and Habitat Quality

Understanding the relationships between group fitness (e.g., reproductive success, group size, adult survival) and quantity and quality of foraging habitat is key to formulating appropriate foraging guidelines for red-cockaded woodpeckers. However, the highly altered state of available resources diminishes our ability to see effects of habitat on group fitness and to determine an optimal amount of foraging habitat. Also, at least two other factors are important to group fitness: presence of helpers (Lennartz *et al.* 1987, Walters 1990, Neal *et al.* 1993a, Beyer *et al.* 1996) and increasing age and experience of breeders (Lennartz *et al.* 1987, Walters 1990, DeLotelle and Epting 1992) are known to increase reproduction. Finally, habitat effects are hard to identify because sample sizes are low, in number of groups studied and/or number of years with which group fitness is estimated. Substantial variation in reproduction can be attributed to stochastic environmental events (e.g., Neal *et al.* 1993a), which can mask other effects in small samples. Despite constraints of available habitat, confounding effects of other factors, and low power due to small samples, important progress has been made in determining effects of habitat quality on fitness.

Several aspects of foraging habitat may affect group fitness. First, territory or home range size has been shown to affect group size and/or reproduction in some populations (DeLotelle and Epting 1992, Hardesty *et al.* 1997, USFWS 1985) but not in others (James *et al.* 1997, Walters *et al.* 2000). For two studies reporting an influence of home range/territory size on fledgling production, much of the effect appears to have

come from whole brood loss or failure to nest (DeLotelle and Epting 1992, Hardesty *et al.* 1997). This suggests that there is a threshold home range size below which reproduction becomes difficult, and it is possible that studies not showing this effect did not sample below the threshold. Home range size for successfully and unsuccessfully nesting groups in northwest Florida averaged 126.3 and 72.4 ha (312 and 179 ac) respectively (Hardesty *et al.* 1997); a threshold home range size for this population under current habitat conditions would fall between these two estimates.

Effects of home range size on fitness vary, of course, with the quality of foraging resources. This point is best illustrated by the large, healthy groups on home ranges averaging only 46.9 ha (116 ac) in the fire-maintained, old-growth longleaf forest of the Wade Tract, GA (including considerable overlap among home ranges, Engstrom and Sanders 1997). These groups have the smallest average home range and the highest average group size and reproduction yet reported (average group size 3.0 to 3.6; average fledglings from successful nests 2.3 to 2.5; Engstrom and Sanders 1997). In addition, effects of foraging habitat on group fitness may interact with the general health of the population. For example, Conner and Rudolph (1991b) reported that loss of foraging habitat affected group size in small isolated populations but not in larger populations.

Recent research has revealed that fire history of the foraging habitat strongly affects group fitness in several different ways. Group size and/or reproduction is negatively affected by dense stands of pines (Hardesty *et al.* 1997, James *et al.* 1997, in press, Walters *et al.* 2000), positively related to percent of ground covered by wiregrass (*Aristida spp.*) or forbs (Hardesty *et al.* 1997, James *et al.* 1997), and negatively related to increasing hardwood midstory (Walters *et al.* 2000). At Eglin Air Force Base in Florida, reproduction was negatively affected by pine density above 16.1 sq. m of basal area per ha (70 sq. ft per acre). Similarly, group size in the North Carolina Sandhills was negatively affected by density of pines less than 35 cm dbh (14 in; Walters *et al.* 2000). Frequent fire increases the quality of foraging habitat in several ways: it provides an open structure by reducing density of overstory and midstory pines and hardwoods, it encourages grass and forb groundcovers, and it may also increase nutrient cycling through the ecosystem and the nutrient content of prey (James *et al.* 1997).

Finally, group fitness increases with increasing numbers of old trees in the foraging habitat. In Louisiana, density of groups, group fitness, and the number of old-growth trees (90 to 120 years in age) were all strongly positively related (Conner *et al.* 1999). In Texas, group size increased with increasing area of pines greater or equal to 60 years in age both within 400 meters of the cluster (Conner and Rudolph 1991b) and at a larger, regional scale (520 to 5200 ha, Rudolph and Conner 1994). Similarly, in central North Carolina group size increased with increasing density of flat-tops (very old pines) in home ranges (Walters *et al.* 2000). Effects of habitat quality on group size are of utmost importance, because of stabilizing effects of helpers on population dynamics, the increase in reproduction in larger groups, and decrease in groups consisting of solitary males.

Other studies have not found a relationship between group fitness and the amount and quality of foraging habitat as measured by traditional variables such as the number and basal area of pines greater than 25 cm (10 in) dbh (Hooper and Lennartz 1995, Beyer *et al.* 1996, Wigley *et al.* 1999, R. Hooper, unpublished).

At the present time, we recognize that fitness of woodpecker groups increases if they have substantial amounts of foraging areas that are burned regularly and have little or no hardwood midstory, an abundant grass and forb groundcover, low densities of small and medium-sized pines and higher densities of large old pines. Again, these results are consistent with those from studies of tree selection, patch selection, stand selection, and home range/habitat quality relationships (see above).

Geographic Variation in Foraging Habitat

There is substantial geographic variation in habitat occupied by red-cockaded woodpeckers. Historically, longleaf pine ecosystems were the most common habitat type and still support most of the largest remaining populations (Carter 1971, Hooper *et al.* 1982, James 1995, Engstrom *et al.* 1996). Within these longleaf pine habitats there is variation in structure and species composition according to soil type and moisture. Red-cockaded woodpeckers also exist in other habitat types including the shortleaf pine communities of Arkansas, Oklahoma, and Kentucky (Wood 1983, Masters *et al.* 1989, Kelly *et al.* 1993, Hines and Kalisz 1995, Zenitsky 1999), the transitional zones of the Piedmont (Steirly 1957), the pond pine communities of eastern North Carolina (J. Carter III, pers. comm.), and the native hydric slash pine system of south Florida (Beever and Dryden 1992). Despite natural geographic variation in habitats, the basic ecology of red-cockaded woodpeckers remains unchanged throughout their range: red-cockaded woodpeckers select old pines in open stands for nesting and foraging, and the open structure that characterizes nesting and foraging habitat is best maintained by frequent, growing season fire.

Variation in Longleaf Pine Communities

Species composition and structure of longleaf pine communities vary according to interacting moisture, soil, and fire factors. Frequently burned sites with deep sandy soils support what are variously known as sandhill, high pine, or xeric sand communities. These xeric sand communities are found throughout the southeast, on alluvial sands, recently exposed terraces, and relict dunes of the entire Coastal Plain as well as along the fall line that marks the transition between Coastal Plain and Piedmont in the Carolinas and Georgia. Two distinct longleaf ecosystems occur on these deep sandy soils: xeric and subxeric longleaf pine woodlands (Peet and Allard 1993, Christensen 2000). Xeric longleaf pine woodlands are characterized by widely scattered longleaf pines, a sparse midstory of turkey (*Quercus laevis*) and bluejack oaks, and sparse groundcovers dominated by wiregrasses (*Aristida stricta* north of the Congaree/Cooper rivers in South Carolina and *A. beyrichiana* to the south, Peet 1993). Within this xeric woodland type, five series have been identified (Peet and Allard 1993): fall line, Atlantic, and southern (Gulf) xeric longleaf woodlands, and Atlantic and Gulf maritime longleaf woodlands.

Subxeric longleaf pine woodlands contain the above species as well as many more that are adapted to somewhat moister conditions (Christensen 2000). This ecosystem type dominated much of the Coastal Plain uplands prior to European settlement (Ware *et al.* 1993, Christensen 2000). Peet and Allard (1993) identified three series within the subxeric ecosystem type: fall line, Atlantic, and Gulf subxeric longleaf pine woodlands.

Mesic longleaf pine communities include flatwoods and savannahs, which differ from each other mainly in structure. Savannahs are characterized by an open canopy and grass groundcover, whereas flatwoods have a somewhat denser canopy and a midstory of shrubs and subcanopy trees (Christensen 2000). The primary cause of variation between flatwoods and savannahs is interacting effects of fire and soil moisture (Peet and Allard 1993). There is no generally accepted classification of these mesic longleaf pine communities (Christensen 2000). Southern flatwoods include saw palmetto (*Serenoa repens*), gallberry-fetterbush (*Ilex glabra*-*Lyonia lucida*), and fern phases. If burned more frequently, these flatwoods may become more like savannahs (Christensen 2000). Longleaf pine savannahs in the Atlantic and Gulf regions contain many endemic species (Peet and Allard 1993, Walker 1993, Christensen 2000), and species diversity for these community types is among the highest in North America (Walker and Peet 1983).

All of these longleaf community types can support red-cockaded woodpeckers if sufficient old-growth and mature pines are available for cavity trees. However, researchers have suggested that in some locations, such as sites of low productivity, extremely dry or wet locations, red-cockaded woodpeckers may need more foraging habitat than those in mesic habitats (Hardesty *et al.* 1997, DeLotelle *et al.* 1987, 1995). These researchers have observed very large home ranges in some locations, possibly because arthropods are limited by sparse groundcovers or low pine density. Expansion of home range size in these habitat types may be a response to low site productivity or a result of past alteration of the forest through overharvest or fire suppression. Low site productivity can also affect how an ecosystem recovers following alteration (Provencher *et al.* 1997, 1998). Whether the effect is natural or human-induced, some populations of red-cockaded woodpeckers in wet or very dry sites are using more foraging habitat. Further research is required before we fully understand how differences in longleaf pine community types influence the foraging ecology of red-cockaded woodpeckers.

Shortleaf Pine Communities

Shortleaf pine communities supporting red-cockaded woodpeckers are found in the Ouachita Mountains of Arkansas and Oklahoma (McCurtain County Wilderness Area and Ouachita National Forest), and on the western edge of the Cumberland Plateau in Kentucky (Daniel Boone National Forest). Shortleaf pine communities are fire maintained, with a two-layered structure of pine overstory and diverse bunchgrass groundcover much like those of longleaf communities. However, unlike most longleaf systems, shortleaf pine communities supporting red-cockaded woodpeckers are in regions of complex topography (Masters *et al.* 1989, 1995, Kalisz and Boettcher 1991, Hines and Kalisz 1995, Zenitsky 1999). These rugged areas have steep and narrow ridges, and communities dominated by shortleaf pine are confined to slopes of southern and western

exposure and to the ridgetops (Masters *et al.* 1989, Foti and Glenn 1991, Kalisz and Boettcher 1991). Mesic sites such as drainages and north-facing slopes are typically dominated by white oak (*Quercus alba*) and some maples (*Acer* spp.; Masters *et al.* 1989, Foti and Glenn 1991).

Historic shortleaf pine/bunchgrass communities have sustained massive intrusion by hardwoods as a result of fire suppression and exclusion, and this intrusion has caused precipitous declines of red-cockaded woodpeckers in these regions (Masters *et al.* 1989, 1995). Return intervals of fire prior to European settlement have been estimated as 3 to 6 years for shortleaf pine ecosystems in rugged terrain (Masters *et al.* 1995). Reintroduction of fire, using a prescribed burning program patterned after the precolonial fire regime, is vital to the survival and recovery of red-cockaded woodpeckers in these regions (Masters *et al.* 1989, 1995).

Several studies indicate that foraging behavior of red-cockaded woodpeckers in shortleaf habitat is similar to that of woodpeckers on the coastal plain. Woodpeckers foraging on shortleaf pines select large old trees in patches that have less hardwood midstory than the surrounding forest (Murphy 1982, Doster and James 1998, Zenitsky 1999). One study of the critically endangered population in Kentucky reported a preference for hardwoods as foraging substrate, for 2 of 5 groups during the 1991 nesting season only (Hines and Kalisz 1995). However, further research in this population showed no such effect (Zenitsky 1999). Again, the severe decline of red-cockaded woodpeckers in Kentucky and other shortleaf habitats is directly related to hardwood encroachment (Masters *et al.* 1989, 1995), and their foraging behavior does not appear to differ from red-cockaded woodpeckers elsewhere in the range (Murphy 1982, Doster and James 1998, Zenitsky 1999).

Red-cockaded woodpeckers can tolerate some overstory hardwoods in foraging habitat, and even in clusters if more than 15.2 m (50 ft) from cavity trees. Inclusions of xeric hardwood species such as post, blackjack (*Quercus marilandica*), and other oaks (*Quercus* spp.), especially in shortleaf forests, are natural components of the ecosystem and do not need to be totally removed for woodpecker management. However, such hardwoods must remain a minor component overall. In the shortleaf forests of Oklahoma, precolonial density of hardwoods was an estimated 4.6 to 5.7 sq. m of basal area per ha (20 to 25 sq. ft per acre; Masters *et al.* 1995). Such densities should be considered maximum for red-cockaded woodpecker management. Estimated pine basal area of precolonial Oklahoma is similar to that of longleaf forests, at 8.0 sq. m per ha (35 sq. ft per acre; Masters *et al.* 1995).

Pond Pine Communities

The remaining pond pine communities that support red-cockaded woodpeckers are found primarily in northeastern North Carolina (J. H. Carter III, pers. comm.). Pond pines were likely sparsely distributed in the upland shrub bogs known as pocosins, but fire suppression has led to increased pine density and hardwood encroachment. Foraging requirements of red-cockaded woodpeckers in this habitat type have not been studied at

all. Management of woodpeckers in pond pines is complicated by the catastrophic nature of the natural fire regime, dangerous accumulation of fuels during years of fire suppression, southern pine beetle outbreaks, and high rates of cavity enlargement by pileated woodpeckers (J. H. Carter III, pers. comm.). Reintroduction of fire is required for continued survival and recovery of woodpeckers in these habitats, but further research is necessary to determine best methods of prescribed burning and foraging habitat requirements.

South Florida Slash Pine Communities

Native slash pine communities support red-cockaded woodpeckers in south Florida (Beever and Dryden 1992). This subspecies of slash pine (*Pinus elliotti* var. *densa*) is the only native pine in this region and is similar to longleaf in both appearance and fire resistance. Similar to longleaf pine, native slash pine has a grass stage and large taproot. Much of the native slash used by red-cockaded woodpeckers is in hydric communities (Beever and Dryden 1992). It may be that slash pine replaces longleaf pine in this region because it can better tolerate very wet conditions.

For red-cockaded woodpeckers, native slash pine habitats differ from those further north in that the pines are generally smaller and may be more sparsely distributed (Patterson and Robertson 1981, Beever and Dryden 1992, Landers and Boyer 1999). The largest size that south Florida slash pines achieve, even in old growth woodlands, is typically 20 to 30 cm (8 to 12 in). Cavity trees in this habitat type are much smaller than normally found in other habitats (Beever and Dryden 1992, Bowman and Huh 1995). However, the presence of fire and old trees in both nesting and foraging areas are critically important here as elsewhere.

Woodpeckers in native slash pine have not been well-studied. Preliminary research has indicated that home ranges of birds in native slash pine are larger than those in other habitats (Patterson and Robertson 1981, Beever and Dryden 1992), but the relationship between habitat requirements and habitat quality has not been investigated in this forest type. Thus, it is not known whether larger home ranges in south Florida result from degraded habitat, natural differences in habitat quality, population density, or even lack of cavity trees. Although further research is necessary to determine the cause of large home ranges in south Florida, results from studies elsewhere suggest that as habitat quality increases, the size of these home ranges will decrease. It is likely that, as pine density, age, and herbaceous groundcovers of south Florida slash pine woodlands increase, resident woodpeckers will still require more foraging habitat than woodpeckers in most other regions but less than they appear to be using at the present time.

Previous Guidelines

Previous guidelines for management of foraging habitat (USFWS 1985, Henry 1989) emphasized the number of pines greater than 25.4 cm (10 in) dbh that should be provided each group of woodpeckers, in stands meeting some broad criteria (e.g., overstory hardwoods 50 percent or less of canopy tree basal area, pines 30 years in age or

greater). These guidelines were important and useful in several ways: the guidelines provided much-needed protection against overharvest of pines; they stressed that red-cockaded woodpeckers require a large quantity of land and they furnished this large quantity of land fairly successfully; and they represented the best estimate of foraging requirements available from research at that time. However, these guidelines were also flawed in some ways: the actual number of pines recommended was based on one population and a small sample ($n=18$); the guidelines may have encouraged high densities of small and medium sized pines now known to be detrimental; and most importantly, researchers have been unable to detect any relationship between the total number or total basal area of pines greater or equal to 25.4 cm (10 in) dbh within a group's foraging area and measures of fitness such as group size or reproduction (e.g., Hooper and Lennartz 1995, Beyer *et al.* 1996, Wigley *et al.* 1999, R. Hooper unpublished). This continued failure to find any relationship between fitness and total number of small and medium sized pines strongly suggests that these variables are not the best way to measure quality or quantity of foraging habitat.

This last point – the lack of relationship between number of pines greater than 25.4 cm (10 in) dbh and group size and/or reproduction—is shown clearly in an analysis recently performed by R. Hooper (unpublished), combining data from nine data sets for a total of 198 groups with mean group size greater or equal to 2 adults. In only two of the data sets did mean number of pine stems greater or equal to 25.4 cm (10 in) dbh approach the standard of 6350 pines set by the 1985 Recovery Plan (USFWS 1985), and one of those data sets determined the original standard. With one exception (Hooper and Lennartz (1995) lacked habitat data for individual groups), these data were pooled for regression analyses of number of pine stems greater or equal to 25.4 cm (10 in) dbh against mean fledglings produced and mean group size. These regressions were significant or nearly significant, but they explained a trivial amount of the variation in independent variables (mean fledglings: $df = 1, 196$; $R^2 = .02$; $P < 0.05$; mean group size: $df = 1, 179$; $R^2 = .04$; $P < 0.01$). Thus, number of young fledged and group size were at best weakly related to the number of pine trees ≥ 25.4 cm (10 in) dbh available to the various groups, and unspecified factors accounted for 98 percent of the variation in number of young fledged and 96 percent of the variation in the group size. Thus, number of pines greater or equal to 25.4 cm (10 in) dbh is not a particularly good measure of foraging habitat requirements.

Implications for Management

Supplying good quality foraging habitat is a critical aspect of red-cockaded woodpecker recovery, especially over the long term, as immediate threats from cavity and cluster limitation are reduced. Our understanding of what constitutes good quality foraging habitat comes from a synthesis of research into selection of foraging habitat and effects of habitat characteristics on group fitness.

Both habitat selection and group fitness are influenced by the structure of the foraging habitat. Important structural characteristics include (1) healthy groundcovers of bunchgrasses and forbs, (2) minimal hardwood midstory, (3) minimal pine midstory, (4) minimal or absent hardwood overstory, and (5) a low to intermediate density of small and

medium sized pines, and (6) a substantial presence of mature and old pines. Thus, the quality of foraging habitat is defined by habitat structure. Although geographic variation in habitat types exist, these structural characteristics of good quality habitat remain true for all geographic regions and habitat types. Previous guidelines stressed quantity of foraging habitat, as defined by number of medium and large trees. Here we expand this emphasis to include habitat quality, as defined by habitat structure, and use area metrics to address quantity. Red-cockaded woodpeckers require foraging habitat that is suitable in both quantity and quality.

Quantifying habitat structure (and thus habitat quality) is more complex than simply requiring a given amount of habitat or number of trees, because habitat structure is measured by multiple variables. Guidelines for foraging habitat (see 8G) are based on the quantification of structural characteristics to the best of current abilities. Frequent fire can facilitate the restoration and maintenance of all but one of these structural characteristics (mature and old pines), and may provide further benefits by increasing the availability of nutrients. In addition, appropriate silvicultural methods will protect, throughout the landscape, the mature and old trees on which red-cockaded woodpeckers thrive.

F. COMMUNITY ECOLOGY:

CAVITY KLEPTOPARASITISM, CAVITY ENLARGEMENT, AND PREDATION

Red-cockaded woodpeckers are a keystone species of fire-maintained southern pine ecosystems because the cavities they create influence the presence or abundance of a suite of cavity-dwelling species in an otherwise cavity-poor environment (Rudolph *et al.* 1990a, Conner and Rudolph 1995a). Excavation of cavities into live pines by red-cockaded woodpeckers requires a relatively long period of time (Jackson *et al.* 1979, Conner and Rudolph 1995a, Harding 1997). Thus, these cavities are in high demand (Dennis 1971a, Harlow and Lennartz 1983, Rudolph *et al.* 1990b, Loeb 1993, Conner *et al.* 1996). Approximately 26 species of vertebrates are known to use cavities excavated by red-cockaded woodpeckers (Baker 1971b, Beckett 1971, Dennis 1971a, Hopkins and Lynn 1971, Jackson 1978a, Harlow and Lennartz 1983, Rudolph *et al.* 1990a, Loeb 1993, Kappes and Harris 1995, Conner *et al.* 1996, Loeb and Hooper 1997). The majority of these vertebrates use either enlarged (below) or abandoned cavities, but red-bellied woodpeckers, red-headed woodpeckers, eastern bluebirds, and southern flying squirrels use normal, unenlarged cavities that red-cockaded woodpeckers could also use. Southern flying squirrels are generally the most commonly observed species in red-cockaded woodpecker cavities other than red-cockaded woodpeckers (Rudolph *et al.* 1990a, Loeb 1993, Kappes and Harris 1995, Laves and Loeb 1999, Mitchell *et al.* 1999), although these observations were made during daylight hours. Eastern bluebirds were more common than flying squirrels in coastal South Carolina (Loeb and Hooper 1997).

Cavity Kleptoparasitism

If a cavity created and used by red-cockaded woodpeckers is usurped by another species, the interaction between that species and red-cockaded woodpeckers is termed cavity kleptoparasitism (Kappes 1997). Until recently, authors have referred to this interaction as cavity competition (e.g., Ligon 1970, Jackson 1978a, Carter *et al.* 1983, Rudolph *et al.* 1990a, Loeb 1993, Kappes and Harris 1995), but the term cavity kleptoparasitism is more correct (Kappes 1997). As Kappes (1997) explains, competition describes an interaction in which both species exhibit a negative effect from the presence of the other. Because cavity usurpers are acquiring a limited resource created by another species, the interaction provides benefits for the usurping species and negative effects on red-cockaded woodpeckers. Kleptoparasitism is the appropriate term for such a positive-negative relationship.

Red-bellied Woodpeckers

Red-bellied woodpeckers are a common cavity kleptoparasite of red-cockaded woodpeckers (Neal *et al.* 1992, Kappes 1997). Usurpation of cavities by red-bellied woodpeckers may result in open roosting for red-cockaded woodpeckers. For example, Kappes (1997) observed 15 adults open roosting during a winter in Florida; 14 of these 15 had suffered loss of cavities to red-bellied woodpeckers. However, how much open roosting may affect survival or territory occupancy is not yet known. Rates of kleptoparasitism by red-bellied on red-cockaded woodpeckers may vary inversely with habitat quality (F. James, unpublished). Similarly, red-cockaded woodpeckers in optimal habitat are likely to suffer less impact from each usurpation event. Thus, increasing the overall quality of the habitat for red-cockaded woodpeckers may be an effective means of controlling effects of cavity usurpation by red-bellied woodpeckers. Retention of snags and provision of nest boxes may reduce effects of red-bellied woodpeckers as well (below).

Southern Flying Squirrels

Reported rates of occupancy of red-cockaded woodpecker cavities by southern flying squirrels range from 9 to 34 percent (Dennis 1971a, Rudolph *et al.* 1990a, Loeb 1993, Laves and Loeb 1999, Mitchell *et al.* 1999). Southern flying squirrels prefer active cavities with non-enlarged entrance tunnels over those with entrance tunnels enlarged (Rudolph *et al.* 1990a, Loeb 1993), and cavity inserts over natural cavities (Lotter 1997). From among active cavities, southern flying squirrels prefer cavities with enlarged chambers over those with regular chambers (Rossell and Gorsira 1996).

Southern flying squirrels could potentially affect red-cockaded woodpeckers through usurpation of cavities or through predation. There is disagreement among researchers over whether cavity usurpation has any negative effects. Some suggest that cavity usurpation lowers nest attempts (Loeb and Hooper 1997), but others have found no evidence that the presence or abundance of southern flying squirrels increases open roosting or decreases nest attempts (Rudolph *et al.* 1990a, Conner *et al.* 1996, Laves

1996, Mitchell *et al.* 1999). Whether or not flying squirrels are significant predators of red-cockaded woodpecker nests is discussed below.

It has been suggested in the past that southern flying squirrels increase with increasing hardwood midstory (Conner and Rudolph 1989, Loeb *et al.* 1992). Yet, Conner *et al.* (1996) observed regular use of red-cockaded woodpecker cavities by southern flying squirrels in loblolly-shortleaf pine habitat with and without hardwood midstory and in open longleaf pine habitat that was nearly devoid of hardwood vegetation. Southern flying squirrels are abundant and ubiquitous, and at the present time the influence of plant species composition and vegetative structure on flying squirrel distributions is not understood.

Reducing Impacts from Cavity Kleptoparasites

The availability of snags may reduce impacts of potential cavity kleptoparasites on red-cockaded woodpeckers. Rates of cavity kleptoparasitism appear to be inversely related to the density of snags within clusters (Harlow and Lennartz 1983, Kappes and Harris 1995). Placement of nest boxes within cavity tree clusters may have a similar effect of lowering use of red-cockaded woodpecker cavities by other species (DeFazio *et al.* 1987, Loeb and Hooper 1997). Thus, provision and maintenance of snags within 230 m (750 ft) of woodpecker clusters should be an important part of a management program to reduce cavity kleptoparasitism. In addition, maintenance of optimal conditions in both nesting and foraging habitat (see 2D, 2E) may decrease impacts of kleptoparasites on red-cockaded woodpeckers.

Cavity Enlargement

Enlarged cavities are those whose entrance tunnels have been widened by one of several species of woodpeckers (Conner *et al.* 1991a, Neal *et al.* 1992). Cavity enlargement is generally done by pileated woodpeckers, but red-bellied and red-headed woodpeckers and northern flickers also enlarge cavities created by red-cockaded woodpeckers (J. H. Carter III, pers. comm.). Pileated woodpeckers greatly expand entrance tunnels and can also enlarge the cavity chamber if sufficient heartwood is present (Conner *et al.* 1991a). Over a period of thirteen years in the Angelina National Forest in eastern Texas, pileated woodpeckers enlarged 41 percent (114 of 276) of unprotected natural red-cockaded woodpecker cavities (Saenz *et al.* 1998).

Cavity enlargement by pileated woodpeckers can have strong negative impacts on individual red-cockaded woodpeckers and, more importantly, on the entire population. Red-cockaded woodpeckers will abandon their clusters if damage to cavities by pileated woodpeckers is great. However, the enlarged cavities created by pileateds provide important habitat for many other relatively large secondary cavity users, such as American kestrels (*Falco sparverius*), eastern screech owls (*Otus asio*), and fox squirrels (*S. niger*; Conner *et al.* 1997b, Saenz *et al.* 1998). In fact, just as red-cockaded woodpeckers are the primary source of cavities for other similar-sized cavity users, pileated woodpeckers are key to the availability of cavities for large cavity-nesting

species (Saenz *et al.* 1998). Therefore, the challenge to management is to reduce the effects of cavity enlargement on red-cockaded woodpeckers without overly impacting large cavity-nesting species of concern.

Why pileated woodpeckers enlarge cavities is unknown. Enlarged cavities are rarely used by pileated woodpeckers for roosting or nesting (Conner *et al.* 1996). Saenz *et al.* (1998) suggest that pileated woodpeckers are attracted to trees bearing signs of woodpecker excavation, but that heavy resin flow often prevents complete nest excavation. Damage by pileated woodpeckers decreases with increasing availability of snags in the general area (Saenz *et al.* 1998), just as rates of cavity kleptoparasitism may decrease with increasing snags. Thus, managers should retain snags throughout lands managed for red-cockaded woodpeckers and consider their protection during prescribed burns.

Cavity damage by pileated woodpeckers may also be related to human disturbance. Initial attempts at midstory control within the cluster may attract pileated woodpeckers if midstory outside the cluster is excessive (J. H. Carter III, pers. comm.). Again, restoration of high quality habitat for both foraging and nesting may reduce impacts from pileated woodpeckers.

Cavity Restrictors

Metal plates that restrict the entrance diameter of red-cockaded woodpecker cavities (Carter *et al.* 1989) can be used to rehabilitate some currently unsuitable cavities or to prevent the enlargement of currently suitable cavities (see 3B). Although these plates may prevent further damage by larger species of woodpeckers, they will not deter the use of cavities by southern flying squirrels or other small species of woodpeckers. When cavity availability is limited (less than four suitable cavities per group or less than one suitable cavity per group member) and enlargement by pileated woodpeckers is common, use of cavity restrictors is absolutely essential to protect existing cavities from enlargement and rehabilitate cavities with minor to moderate entrance enlargement. Use of restrictors to prohibit use of cavities by red-bellied woodpeckers is not recommended (see 3B).

Restrictors require careful monitoring on an annual basis, to ensure that negative effects on red-cockaded woodpeckers are minimized (see 3B). For this reason, their use must be judicious rather than haphazard or wholesale. In addition, enlarged cavities that have been abandoned for several years should not be restricted or should have any existing restrictors removed, so that they may be available to secondary cavity nesters. Similarly, if cavities are not limited, then restrictors are not necessary and some enlarged cavities can be tolerated.

Predation

Rat Snakes

Red-cockaded woodpeckers excavate resin wells around cavity entrances to create a coat of fresh resin, typically extending several meters below and above the entrance and occasionally to the ground. They also scale loose bark from the bole of the cavity tree and nearby pines. During the 1970's, several biologists realized that these behaviors serve to protect the nests against predation by rat snakes (Ligon 1970, Dennis 1971b, Jackson 1974, 1978a), and in the late 1980's Rudolph *et al.* (1990a) documented experimentally the effectiveness of the resin barrier against climbing rat snakes.

Rat snakes are excellent tree climbers (Jackson 1976) and frequently prey on cavity-nesting birds (Fitch 1963, Jackson 1970). They attempt to climb cavity trees and cavity trees with nests more often than expected by chance alone, evidence that rat snakes are able to detect which trees contain cavities and also which cavity trees contain nests (Neal *et al.* 1993b). Sometimes, rat snakes are able to breach the resin barrier and prey on cavity contents such as eggs, nestlings, or even adults (Jackson 1978a, Neal *et al.* 1993b, 1998).

However, reports of individual predation events by rat snakes on red-cockaded woodpeckers are relatively scarce, and there is no evidence that such predation affects woodpeckers at the population level. For example, there was no difference in average reproduction between nests in cavity trees fitted with snake exclusion devices and untreated cavity trees over three years in the longleaf pines of northwest Florida (L. Phillips, unpublished). It is likely that the resin barrier is a highly effective means of deterring rat snakes, especially in longleaf pine.

Southern Flying Squirrels

Although flying squirrels are known to eat eggs of red-cockaded woodpeckers on occasion (Harlow and Doyle 1990), there is little consistent evidence that flying squirrels significantly depress reproduction of red-cockaded woodpeckers. Two experimental studies have been conducted comparing reproductive success of red-cockaded woodpeckers in clusters with and without squirrel removal (Laves and Loeb 1999, Mitchell *et al.* 1999). Laves and Loeb (1999) reported lowered reproduction in clusters without squirrel removal, resulting from increased whole brood loss in one year and increased partial brood loss in the following year. Mitchell *et al.* (1999) reported no difference in overall reproduction between clusters with and without squirrel removal, but noted increased partial brood loss in clusters that had squirrels removed. In addition, Conner *et al.* (1996) did not detect any relationship between abundance of southern flying squirrels and reproductive success of red-cockaded woodpeckers in eastern Texas. No study has yet shown an effect of flying squirrels on red-cockaded woodpeckers at the population level (Mitchell *et al.* 1999). Thus, it appears that impacts of flying squirrels on red-cockaded woodpeckers are not strong, at least in the populations in which they have been assessed.

Implications for Management

In general, predator control is not an effective method of achieving stabilization or increases in bird populations, because predators rarely regulate population size (Côté and Sutherland 1997). For red-cockaded woodpeckers, predators were not among the original causes of decline, and their removal will not result in population increases. Only habitat restoration, including prescribed burning, protection of mature and old-growth trees, and cavity provisioning, can stabilize and increase populations by removing the original causes of decline.

Critically small populations, however, may not be able to withstand the loss of an occasional nest to predation by southern flying squirrels or rat snakes. For these populations, the use of squirrel and rat snake exclusion devices (see 3C) may be considered, but should not take the place of more fundamental management. Such methods are not appropriate in larger populations, because they may cause unintentional harm and can focus attention and resources away from habitat management and restoration.

In contrast, cavity enlargement by pileated woodpeckers can have population-level effects in even fairly large populations by causing cluster abandonment. Restrictors (see 3B) are an essential management tool to be used judiciously in appropriate circumstances, with proper maintenance. Whether cavity kleptoparasitism by red-bellied woodpeckers negatively affects red-cockaded woodpecker populations requires further study. Effects of cavity kleptoparasitism by flying squirrels are under debate but are not considered strong or consistent enough to warrant flying squirrel removal or exclusion except perhaps in critically small populations (less than 30 potential breeding groups). Provision of nest boxes is a non-invasive technique that may help reduce effects of cavity kleptoparasitism (Loeb and Hooper 1997). Some evidence suggests that any effect of red-bellied woodpeckers (F. C. James, pers. comm.) and southern flying squirrels (Loeb and Hooper 1997) may increase with habitat degradation. In general, maintaining good quality nesting and foraging habitat (see 8D, 8G), providing sufficient numbers of suitable, unenlarged or restricted cavities (8C), and retaining snags in the landscape are the best management tools to reduce possible effects of occasional predation and cavity kleptoparasitism and to control the far more serious impacts from cavity enlargement.

G. THE ROLE OF FIRE IN SOUTHERN PINE ECOSYSTEMS

Fire is an integral component of the southern pine/bunchgrass ecosystems of the southeastern United States, and fire suppression is a principal factor in the decline of these ecosystems and characteristic species such as red-cockaded woodpeckers (see 1A). In this section, we review the history of fire in the region and the fire dependence of the species comprising southern pine ecosystems. In 3F, we discuss prescribed fire and red-cockaded woodpecker management, including description of ignition techniques, benefits to other species, and concerns about negative impacts. Guidelines for using prescribed fire in the management of red-cockaded woodpeckers are presented in 8I.

History of Fire in the Southeast

Fire is a natural ecosystem component that gained and lost importance in North America as the glaciers retreated and advanced. Pyrophytic vegetation in what is now the southeastern United States evolved in response to fires ignited by lightning long before the last glacial retreat roughly 10,000 years ago (Komarek 1968, 1974, Ware *et al.* 1993). Aboriginal people immigrated into the region during the last glacial period, and so the development and spread of fire-dependent ecosystems as the last glaciers retreated were influenced by both climate and the presence of Native Americans (Delcourt *et al.* 1993, Frost 1993, Ware *et al.* 1993). Modern plant assemblages have remained relatively stable for the past 6,000 years (Webb 1988, Frost 1998), despite some oscillations in fire frequency caused by minor changes in climate (Frost 1998). Thus, the ecosystems in place at the time of European exploration of North America had been in place for thousands of years (Frost 1998), and those in the southeastern region were shaped primarily by fire.

Prior to European colonization, there were few natural firebreaks in the southeast, and so fires burned for extended periods and over large regions. Return intervals for these natural fires were as frequent as 1 to 3 years in much of the Atlantic and Gulf Coastal Plains, and as frequent as 4 to 6 years in Upper Gulf Coastal Plains and the Piedmont (Wahlenburg 1946, Frost 1998). Some areas, such as slopes with northern aspect and wetlands, may have burned at frequencies of 7 to 25 years (Frost 1998).

Fire intensity is intimately related to fire frequency, and together they are a primary determinant of ecosystem structure and species composition. Over much of the southeast, frequent fires were low in intensity, as evidenced by the species adaptations and structure of longleaf and shortleaf communities (below). In some regions, fires were less frequent and of stand-replacing intensity. Such areas support pines that are adapted to stand-replacing fires, such as sand, Table Mountain (*P. pungens*), pitch, and pond pines (Landers 1991). Only the latter two species are used by red-cockaded woodpeckers. Occasionally, some patches of longleaf and shortleaf communities may have undergone stand-replacing fires as a result of unusually long fire intervals. Thus, precolonial longleaf and shortleaf ecosystems were likely mosaics of mostly multi-aged woodlands with occasional even-aged stands (Landers 1991). Community species composition and tree density varied as functions of the fire regime, moisture gradient, and soil fertility.

The relative role of Native Americans in augmenting the lightning fire regime likely varied regionally, depending upon the frequency of lightning fire (Frost 1998). Native Americans may have shifted the seasonality of fire from the lightning season to include fires in fall and winter as well (Higgins 1986, Frost 1998). In general, however, it is not necessary to distinguish the exact contributions of anthropogenic and lightning fire to understand the role of fire in shaping and maintaining the ecosystems of the southeast. Native Americans were an integral component of these developing ecosystems for the 10,000 years of the Holocene.

Like the Native Americans, early European settlers also used fire as a tool, practicing slash and burn agriculture throughout the southeast during the 18th and 19th centuries. Farmers and ranchers continued to use fire to improve grazing quality for free ranging livestock into the first half of the 20th century, setting fires primarily in the early spring (Otto 1986, Frost 1993). As timber surpassed cattle in economic importance, however, fire was increasingly seen as the enemy of the woodland manager. Fire detection and suppression systems were instituted, and large fires became increasingly rare.

Much of the 20th century was a time of active, aggressive fire suppression. Increasing human-made firebreaks such as roads, fields, and powerlines also reduced the extent of natural fires and fire frequency. Prescribed fire was recognized by some as an important tool to reduce the risk of catastrophic wildfire (Sachett 1975) and was occasionally used to improve game habitat (Stoddard 1935), but these fires were set in the winter months. Dormant season fires were not as effective as natural, intense, growing season fire in maintaining the open pine woodlands and savannahs that red-cockaded woodpeckers require. By the 1960's, fire suppression and exclusion threatened the existence of the species.

Fire Dependence and Adaptation

Many species of the southern pine-bunchgrass ecosystems show adaptations to frequent, low intensity fires, including red-cockaded woodpeckers. A fundamental adaptation of red-cockaded woodpeckers to fire is the excavation of roost and nest cavities in live pines, a response to the lack of snags and hardwoods in fire-maintained pine systems (Ligon 1970, Jackson *et al.* 1986). This ability to excavate cavities in live pines is not only important to red-cockaded woodpeckers but also to the many other species that use these cavities in the otherwise cavity poor environment (Brennan *et al.* 1995, Conner *et al.* 1997a; see 2F). Excavation of cavities in live pines has in turn led to the complex and unusual cooperative breeding system of red-cockaded woodpeckers (Walters 1990, Walters *et al.* 1992a; see 2B). A second adaptation of red-cockaded woodpeckers to fire is the abandonment of cavity clusters in the presence of substantial hardwood midstory. This may be a mechanism for avoiding the dangerous fires that will inevitably occur when the midstory is ignited. The severe impact and continuing threat of fire suppression to red-cockaded woodpeckers are discussed in 1A and 1B.

Plants of the southern pine ecosystems are well adapted to and require frequent burning. Many groundcover plants require growing season fires for flowering and fruit and seed production (Platt *et al.* 1988a, Streng *et al.* 1993, Walker 1993). Platt *et al.* (1988a) showed that herbaceous plants undergoing growing season fire not only increased flower production but also increased synchronicity of flowering, facilitating pollination and reducing risk of hybridization. Populations of these herbaceous plants, therefore, are regulated by fire. Ferguson (1998) recounted a typical example of a population of Florida skullcaps (*Scutellaria floridana*) reduced to three individuals which then swelled to over 100 individual plants following a growing season fire. Walker (1993) lists nearly 400 rare, mostly herbaceous plants of longleaf pine communities, of

which over 90 percent are adapted to growing season fire. Diversity of herbaceous plants in longleaf systems place these among the most highly diverse ecosystems in North America (Walker and Peet 1983, Peet and Allard 1993). This diversity is maintained by frequent fire and severely threatened by fire suppression (Christensen 1981, Ware *et al.* 1993, Peet and Allard 1993, Glitzenstein *et al.* 1998b, Walker 1998). Over 120 species of plants associated with red-cockaded woodpecker habitats are currently on the regional list of proposed, endangered, threatened, and sensitive species (USFS 1995).

Pine trees in general are noted for being well adapted to fire, but longleaf and south Florida slash pines in particular are extremely well adapted to fires of high frequency and low intensity (Landers 1991). Adaptations providing these two species with resistance to fire damage include the grass stage of seedlings, a large taproot, special bark characteristics, absence of branches below the crown, and the typical clumped arrangement of needles at the growing tips of branches (Wahlenburg 1946, Landers 1991). Longleaf and south Florida slash pine seedlings maximize taproot growth and minimize early height growth; the reverse is true of loblolly pine (Landers 1991). In addition, fire enhances seed germination and seedling establishment. Finally, both fire-adapted species facilitate the ignition and spread of fire by producing highly resinous, long needles and shedding them frequently (Platt *et al.* 1988b, 1991, Noss 1989, Landers 1991). This facilitation of fire maintains environmental conditions that are beneficial to these species but detrimental to competitors. Through its profound influence on the fire regime, longleaf pine is a key species in the longleaf pine communities (Platt *et al.* 1988b, 1991, Noss 1989, Landers 1991). Fire suppression and the resulting invasion of hardwoods have altered almost all longleaf pine ecosystems (Frost 1993).

Engstrom (1993) reported 36 species of mammals and 86 species of birds (35 permanent residents, 22 winter residents, and 29 breeders) characteristic of southeastern longleaf pine ecosystems. Many of these animals, and many more plant species, are threatened by fire suppression. USFS (1995) reported that 56 animal species associated with red-cockaded woodpecker habitats are currently on the regional list of proposed, endangered, threatened, and sensitive species. In addition, entire associations of species have been affected, such as the threatened gopher tortoise (*Gopherus polyphemus*) and the 13 listed and candidate species of animals that depend on gopher tortoise burrows (USFS 1995). Fire benefits shortleaf pine communities as well, although these have not received as much research attention as longleaf systems. Masters *et al.* (1998) reported that species richness and diversity of small mammals increased in relation to midstory removal and prescribed fire, and no species was adversely affected by fire.

Guyer and Bailey (1993) reported 34 amphibian and 38 reptilian species that are closely associated with longleaf pine forests. Thirty-five percent of the amphibians and reptiles inhabiting longleaf pine forests, and 56 percent of the longleaf pine specialist species, were listed by at least one conservation agency as being of special concern. Fire suppression was identified as a primary cause of the decline of these species.

There is growing evidence that frequent fire may increase arthropod diversity and abundance (Folkerts *et al.* 1993, Provencher *et al.* 1997, Collins 1998). Groundcovers

maintained by frequent fire may support more arthropods than areas with a hardwood midstory (Provencher *et al.* 1997, Collins 1998), although populations of some species, especially those in the leaf litter, may initially decline after burning. Provencher *et al.* (1997) suggests that invertebrate densities may increase following fire since resprouting plant tissue contains higher levels of nitrogen relative to carbon than older tissue (Christensen 1993), thus providing more palatable forage. Nutrient content of arthropods may increase also, following the release by fire of nitrogen and other nutrients into the soil (James *et al.* 1997).

Implications for Management

Fire is an essential element of southern pine ecosystems, critical to the maintenance of habitat for red-cockaded woodpeckers and many other species. Frequent fire has helped to shape and maintain some of the most highly diverse ecosystems outside the tropics. However, natural fire can no longer maintain suitable habitat for red-cockaded woodpeckers and associated species, because the fragmentation of landscapes has reduced fire spread, duration, and therefore fire frequency. Thus, prescribed fire is a fundamental solution to the conservation of red-cockaded woodpeckers and their ecosystems. To maximize benefits, the frequency, intensity, and season of prescribed fire should mimic the historic natural fire regime as closely as possible (see 2G).

3. MANAGEMENT TECHNIQUES

A. POPULATION MONITORING

Population monitoring is a critical component of the conservation and recovery of red-cockaded woodpeckers. Most importantly, monitoring is required to assess population trends and thereby evaluate the effectiveness of management techniques and adjust them accordingly. Fortunately, red-cockaded woodpeckers are more easily monitored than most species because of their conspicuous active cavity trees and the exceptional stability of territory locations. In this section, we first review techniques for cavity tree surveys. We then discuss methods of population monitoring to (1) document population trends, (2) determine success of translocations, (3) measure impacts of certain activities, (4) implement mitigation actions, and (5) conduct research. Guidelines for population monitoring are given in 8B. Many activities conducted during population monitoring, such as climbing cavity trees, using a video probe (“peeper”), and banding, require federal endangered species permits (see Appendix 1) and may require state permits as well.

Surveying for Cavity Trees and Clusters

Accurate inventories of cavity trees and clusters is an essential step of any population monitoring scheme. Such a database is maintained by comprehensive surveys of occupied and potentially occupied habitat at least once every 10 years and annual updates of the activity status of cavities, cavity trees, and clusters. To limit undetected

cavity trees and misjudged activity status, surveys are best performed by trained personnel following specific survey protocol.

In most habitat types, surveys are best conducted by foot, using transects spaced to allow overlapping visual coverage of all potential cavity trees (pines at least 60 years in age, in pine and pine-hardwood stands regardless of tree density). Proper spacing of transects will vary with overstory density, midstory density and height, and terrain.

Aerial surveys, by helicopter or small fixed wing aircraft, can be useful in certain habitats such as pocosin or bays where access by foot is difficult. Such surveys, performed by experienced observers, will locate most clusters containing multiple cavity trees but will rarely detect all cavity trees in a cluster or all clusters. In other words, aerial surveys can document the presence of cavity trees but not their absence. Ground surveys can then be used to verify the results of aerial surveys and to locate all cavity trees in detected clusters.

Monumentation and Monitoring of Cavity Trees

An accurate database of cavity trees also requires careful monumentation and monitoring. Here we describe procedures for monumentation and monitoring that will provide this accurate database.

1. Each cavity tree is marked with a uniquely numbered metal tag, without duplicating tag numbers or reusing tags from dead trees. Cavity trees are also typically painted with a single white band, 10 to 20 cm (4 to 8 in) wide, 1.5 to 1.8 m (5 to 6 ft) above the ground.
2. When a cavity tree is first discovered, the following data are recorded:
 - a. cavity tree number
 - b. cluster affinity
 - c. location (GIS or UTM coordinates)
 - d. tree species
 - e. overstory type
 - f. midstory type and height
 - g. groundcover type
 - h. management actions required
3. For each cavity in a newly discovered tree, or for a newly discovered cavity in a known cavity tree, the following data are recorded:
 - a. stage of excavation
 - b. shape
 - c. height
 - d. orientation
 - e. activity status
 - f. management actions required.

4. The following data are recorded at a convenient time, upon discovery or later:
 - a. dbh, height, and age of the cavity tree
 - b. stand basal area
5. The activity status of all cavities, cavity trees, and clusters are visually inspected and assessed once each year, typically in February through April. At this time the following data are recorded:
 - a. cavity number
 - b. stage of excavation
 - c. shape
 - d. activity status
 - e. site changes not previously noted
 - f. management actions required.
6. All management treatments of cavities and clusters such as use of restrictors or prescribed burns are recorded when performed.

Monitoring to Determine Population Size and Trend

We describe four levels of monitoring in detail below. A comparison of these techniques is presented in Table 1, below. Recommendations for the appropriate level of monitoring based on population size and role in recovery are given in 8B.

Level I

Level I monitoring, the simplest and least labor intensive of all methods, consists of checking the activity of each cluster annually during the breeding season (March – July). It is important that these assessments are performed by an experienced red-cockaded woodpecker biologist, and it is done during the breeding season because woodpecker populations are lowest during this time. Consistency is also important: comparing census data from the breeding season with data collected during the non-breeding seasons will result in false interpretations of population trends.

All clusters active within the last 5 years, and inactive clusters having undergone habitat restoration within that time, are checked for activity. Within each cluster, all cavities that have been active within the last 5 years are evaluated. If all cavities are inactive in a cluster that is normally active, a thorough search for new cavity trees is conducted in suitable habitat within 0.4 km (0.25 mi) of the cluster center.

Level I monitoring is highly dependent on how accurately the clusters are designated (i.e., how well the cavity trees are assigned into clusters). Verifying accuracy of cluster designation requires at least some intense monitoring initially (see Reed *et al.* 1988).

Even when accurately applied, Level I monitoring does not give a definitive estimate of population size or number of potential breeding groups (Table 1). It can detect population increases and decreases once they are well underway but, unlike higher

levels of monitoring, it cannot detect early stages of declines or indicators of impending change. Therefore, Level I monitoring may only be appropriate in non-recovery populations. When this type of monitoring is combined with intensive monitoring of a significant portion of the population (Level IVa, see below), a much better estimate of population size and trend is possible.

TABLE 1. Comparison of techniques for monitoring red-cockaded woodpecker populations.

Level	Effort Required				Estimates Obtained				
	Cluster Update	Nest Checks	Group Checks	Band Birds	Pop. Size	Pop. Trend	#Breeding Groups	Repr. Effort	Repr. Success
I	Yes	No	No	No	Poor	Poor	Poor	No	No
II	Yes	Yes	No	No	Poor	Poor	Fair ¹	Fair ¹	No
III	Yes	No	Yes	No	Fair	Fair	Good ¹	No	No
IV a	Yes	Yes	Yes	Yes	Good	Good	Good	Good	Good
IV b	Yes	Yes	Yes	Yes	Exc.	Exc.	Exc.	Exc.	Exc.

¹Method provides minimal estimate.

Level II

Level II monitoring is the cluster activity checks of Level I combined with nest checks of active clusters to document nesting activity. Nest checks are periodic visits to active clusters during the breeding season. They consist of (1) scraping or pounding on active cavity trees in an effort to flush incubating birds, (2) listening for nestlings begging for food, (3) using a video probe or climbing equipment to inspect potential nest cavities, and/or (4) watching for adults carrying food to a cavity. Nest checks are conducted every 7 to 11 days until a nest is detected, then terminated for that cluster. In Level II monitoring, no birds are banded and no fledgling censuses are conducted.

If Level II monitoring is used, the number of days between nest checks and the month in which nest checks are performed must be consistent across years. Data collected during two cycles of nest checks in May cannot be compared to data collected during April or June, or to four cycles of nest checks.

Level II monitoring gives rough estimates of number of potential breeding groups and the overall breeding effort for a particular year (Table 1). Level II monitoring, unlike Level I, can detect the early stages of population decline, manifested as an unusually low proportion of active clusters in which nesting is detected. However, Level II monitoring will underestimate the number of nests, because early nests that fail and some re-nests will not be detected. In addition, Level II monitoring does not give any information concerning nest failure rates and productivity. Therefore, this monitoring method is best used in combination with intensive monitoring of color-banded birds in a significant number of clusters (Level IVa, below). Data from intensively monitored clusters can then be used to estimate the proportion of groups that do not nest and the rate of nest failure, giving good estimates of productivity and the total number of groups.

Level III

Level III monitoring is the “group checks” method described by USFS (1995). Each group in a sample of clusters is followed for a half an hour to an hour, immediately after the birds exit their cavities in the morning, to determine group size. Group size is determined by observation of bird behavior and groups are classified as (1) two or more birds, (2) solitary bird or (3) no birds. Groups of two or more birds that remain together and peacefully interact are assumed to represent potential breeding groups.

Care must be taken to accurately classify the group. Red-cockaded woodpeckers roosting extra-territorially in clusters occupied by one or more residents, captured clusters, and territorial conflicts can confuse the observer and result in erroneous group classifications. If doubt as to group membership exists, the follow time is extended or the follow repeated on another day. Two observers may be necessary if two clusters are located very close together or if cavity trees within a cluster are spread over a large area.

Group checks are valid only if implemented during the breeding season. Groups of two or more birds at other times of the year may or may not represent potential breeding groups. The method is labor intensive (one group per observer per day at best) and complete population censuses are possible only in small populations or with multiple observers. Level III monitoring yields fair information on population size and trend and good information on the number of potential breeding groups, depending on the sample size (Table 1). It yields only indirect evidence of reproductive effort and cannot be used to estimate reproductive success.

Level I, II, and III Combined

A combination of Level I, II and III monitoring has considerable advantages over any one technique. Level I cluster checks can be used to determine which sites are active (or inactive) in the weeks immediately prior to the breeding season. Level II monitoring can be used to locate as many nests as possible during the early to mid part of breeding season (egg and nestling stages). A single observer can check a dozen or more clusters a day for nests. Level III monitoring is employed to determine group composition in active clusters where no nesting effort is documented. Collectively, these three methods allow sampling of much larger sample sizes with the same amount of personnel as Level III monitoring alone. Estimates of population size and trends, the number of potential breeding groups, and reproductive effort can be good as long as monitoring methods and effort remain constant across years. However, these methods yield no information on reproductive success.

Level IV

Level IV monitoring consists of the cluster activity checks of Level I monitoring, plus intensive monitoring of a sample of clusters. It may also include Level II and/or Level III monitoring of clusters not included in the sample.

Intensive monitoring consists of color banding all nestlings and adults, checking nests at regular intervals throughout the breeding season, and conducting fledgling and adult censuses. In general, adult red-cockaded woodpeckers in intensively monitored clusters are banded during the non-breeding seasons. Banding of adults during the breeding season is limited to those birds that cannot be captured using normal procedures at other times of the year (e.g., a female with no known roost cavity). Captures are made at roost cavities at sunrise or sunset, and banding does not take place after dark or during wet weather.

Nest checks are conducted on a 7 to 11 day cycle. More frequent checks yield little additional data and subject the birds to unnecessary disturbance. Longer nest check intervals greatly increase the likelihood that nestlings will be too old to band when found and increases the number of failed nests that are undetected.

Nestlings are banded between the ages of 5 to 10 days old. Banding nestlings older than ten days is prohibited because of increased risk of injury and mortality. Data from the North Carolina Sandhills indicate that the probability of fledging decreases as a direct function of age at banding after 11 days of age (M. LaBranche *et al.*, unpublished). Banding nestlings younger than 5 days is not possible because they cannot accommodate 3 color bands on a leg (see below). No further nest checks are conducted once nestlings are banded.

Nestlings and adults are banded with color combinations consisting of three bands (size XB) on one leg and a single color band and the U.S. Geological Survey aluminum band (size 1A) on the other leg. Birds are not to be banded with one or two color bands alone on a leg. Red-cockaded woodpeckers so banded have a higher likelihood of becoming “toe-hung” over time, with the ensuing loss of one or more toes or a foot. Unnatural mortality associated with this condition has been documented (J. Carter III *et al.*, unpublished).

A fledgling census is performed for each banded nest in the sample between 2 and 14 days after the projected fledging date (hatching date plus 26 days); earlier within that period is better. The fledgling census lasts a minimum of one hour and is conducted in the morning when activity is greatest. During the fledgling census, the number of fledglings and their sex are recorded. Sexes are determined by an unobstructed view of a fledgling’s entire crown; females have a black crown and males have a red crown patch. If a fledgling is not detected in the first check, a second census is conducted within the next 10 days. If no fledglings are detected in two censuses, the cluster is checked for a second nest attempt.

Adult red-cockaded woodpeckers are counted and identified whenever encountered during monitoring activities, and special attention is given to identifying adults during fledgling censuses. Unbanded adults, unidentified adults, and those with bands needing cleaning or replacement are scheduled for capture or additional observation after the breeding season.

Level IV monitoring can consist of intensively monitoring a significant sample of a given population (Level IVa) or the entire population (Level IVb). Minimum sample size for Level IV monitoring is 30 groups or 25 percent of all active clusters, whichever is larger (see below). Level IVa monitoring provides good estimates of population size, trend, and reproductive success (Table 1). Level IVb monitoring yields the actual values of these parameters rather than estimates. It is important that populations essential to recovery and populations on public lands be monitored under Level IV. Populations on private lands that are involved in translocation and mitigation are monitored under Level IV as well.

Sampling

In Level I and Level IVb monitoring, all clusters are included. Samples of clusters are monitored in Levels II, III, and IVa. Sampling designs depend on management objectives and data requirements, population size and trend, and the spatial structure and habitat types of the population. A minimum of 25 percent of active clusters or 30 groups, whichever is largest, is sampled annually. This minimum is increased if more data are required, the population is small or declining, and/or if several subpopulations exist. The distribution of sampling is determined by an experienced woodpecker biologist with the aid of a wildlife statistician. No sample will be sufficient to track small changes in population size from year to year precisely (see below). Also, it is critical that monitoring methods not be mixed between years.

Sampling error typically exceeds the annual rate of change in population size in all but the largest populations, but is less than annual variation in productivity for most populations. Because of this, intensive monitoring of a sample of groups (Level IVa) can provide reliable estimates of productivity within a short period, but cannot estimate population trend until used for a longer period of time (e.g., 5 years).

Accuracy of population trend assessment is also dependent on sample size. For example, the proportion of clusters containing solitary males is an indicator of a declining population. To be 95 percent confident that an observed value of 20 percent solitary males is truly above the acceptable level of 10 percent, a sample of 65 clusters is required. However, to determine if an observed value of 15 percent solitary males is greater than 10 percent, a sample of 205 clusters is required. Thus, small samples cannot detect anything but large-scale changes in population trends.

Translocation Monitoring

Translocation is being increasingly used as a management tool to bolster small or fragmented red-cockaded woodpecker populations (see 3D). Determining fate of translocated birds and the impacts of their removal on donor populations is crucial to the evaluation of this management tool. Also, sex and status of red-cockaded woodpeckers within their group of origin must be known prior to removal, which requires prior color banding of the donor group. Birds within and near recipient clusters are color-banded to

determine group composition and needs. The color bands on translocated birds allow for tracking of the individual.

Once a translocated bird is released, no further observations are required until the next breeding season. If follow-up checks on the translocated bird are made, wait at least one week after the release to allow the bird to become accustomed to its new cluster and to form a pair bond. Employ intensive monitoring during the next breeding season to document nesting effort and success. Again assuming positive results, the cluster may be assigned thereafter to a less intensive monitoring category.

Translocation of red-cockaded woodpeckers requires state and federal endangered species and bird banding permits (see Appendix 1). Specific protocols, available from the Red-cockaded Woodpecker Recovery Coordinator, are followed, and all translocation attempts are reported to the Recovery Coordinator through the Annual Monitoring Report process.

Monitoring To Measure Impacts

Documentation of specific impacts on red-cockaded woodpeckers, including those of management actions, requires intensive monitoring. Although some events may impact entire populations (such as hurricanes or fires), monitoring to measure impacts is usually applied to small numbers of clusters subject to a specific activity or all clusters in very small populations (less than 30 potential breeding groups). Examples of activities that may impact red-cockaded woodpeckers are development (e.g., roads, golf courses, housing areas), military training (e.g., impact areas, mechanized training, bivouacs, etc.), and timber management practices (e.g., thinnings, harvests).

Monitoring of impacts includes determination of cavity and cluster activity, nest checks, color banding of nestlings and adults, and fledgling and adult censuses. Such monitoring may be restricted to only those clusters actually affected by a project (a cluster bisected by a road, for example) or it may also encompass the clusters immediately adjacent to the affected cluster(s). This is usually done in the context of assessing incidental take (see 4A) in relation to an activity, although such studies are often inadequate to provide definitive evidence of the cause of losses, especially since some losses may not manifest themselves until years after the initial impact.

Impacts to woodpecker groups are best measured through experimental studies in which treated clusters are paired with control clusters. Such experiments require detailed design by an experienced biologist with thorough knowledge of the subject population, and input from a wildlife statistician is desirable. Simple monitoring of impacted groups, as described above, can only document the continued existence of the group; it cannot be used to identify any effect less than the disappearance of the group. Experiments, on the other hand, may reveal impacts to group size or reproduction.

Mitigation Monitoring

Monitoring is often required to document the effects of implementation of Reasonable and Prudent Alternatives and Reasonable and Prudent Measures pursuant to Section 7 of the Endangered Species Act (Act), implementation of Habitat Conservation Plans pursuant to Section 10 of the Act, and actions taken to offset violations of Section 9 of the Act. In these cases the use and documentation of specified monitoring actions is generally a legal requirement. For further information concerning mitigation, see 4A.

Monitoring for mitigation includes (1) monitoring of those clusters to be impacted as well as nearby clusters, and (2) monitoring of the population containing the mitigation site. The level of monitoring for impacted clusters and those nearby is determined on a case-by-case basis. Monitoring of the population containing the mitigation site is intensive (Level IV), because documentation of the creation of new groups requires comprehensive knowledge of the current distribution of woodpecker clusters and groups within the subject population.

This comprehensive knowledge of the population to contain the mitigation site is needed prior to the installation of artificial cavities. If artificial cavities are placed too close to another group (0.4 km [0.25 mi] or less), the provisioned site is likely to be captured by the adjacent group and no new group will be formed. If artificial cavities are placed too far from other groups (more than 1.6 to 3.2 km [1 to 2 mi]), the likelihood of woodpeckers finding the new site is reduced unless translocation is used.

Comprehensive knowledge of the mitigation site is also necessary for accurate determination of new group formation. Formation of a new group cannot be assumed from simply observing red-cockaded woodpeckers in the provisioned site unless the birds observed are known not to be part of another group. Birds from adjacent groups can be expected to routinely forage around and within the new site and may cross-roost in the new cluster. Mitigation is deemed successful only when monitoring clearly demonstrates that a new group (of equivalent status to the group impacted, solitary male or potential breeding group) has been formed and that it represents a net gain of one group in the area occupied by the provisioned site and all immediately adjacent territories (within 3.2 km [2 mi]). Such determination is only possible through the use of Level IV monitoring.

Research Monitoring

Research monitoring is used to investigate all aspects of the biology of red-cockaded woodpeckers, including, but not limited to, demography, social behavior, and habitat use. Color banding of red-cockaded woodpeckers (Level IV) is often required. Research monitoring that involves handling, banding, or disturbance of red-cockaded woodpeckers requires the appropriate state and federal endangered species and bird banding permits. Depending on the circumstances, a Section 7 consultation or Section 10 (Incidental Take) Permit may be required.

Annual Reporting of Monitoring Results

Managers are required to submit an Annual Red-cockaded Woodpecker Population Data Report to the Red-cockaded Woodpecker Recovery Coordinator containing results of their annual monitoring efforts. Such reporting is a critical aspect of woodpecker management and recovery.

B. CAVITY MANAGEMENT: ARTIFICIAL CAVITIES AND RESTRICTOR PLATES

Loss of cavities and cavity trees was a primary cause of the decline of red-cockaded woodpeckers, and is a substantial threat currently (see 1A, 1B). Today's forests simply do not contain sufficient numbers of mature and old-growth trees for populations to remain stable or increase in the absence of human intervention. Red-cockaded woodpeckers will abandon clusters if insufficient suitable cavities are available. Cluster abandonment leads directly to population extirpation, because populations of red-cockaded woodpeckers are regulated by the number of potential breeding groups rather than by annual variation in reproduction and survival (Walters 1991; see 2B), and because natural formation of new clusters is very slow at least under current conditions (see 2B). Therefore, cavity management through the use of artificial cavities and restrictor plates is absolutely critical to the conservation of most populations.

Cavity ecology, including reasons why the birds need mature and old-growth trees, is discussed in 2D. Community ecology, including the use and enlargement of red-cockaded woodpecker cavities by other species, is discussed in 2F. In this section, we describe the various methods of artificial cavity installation and their respective advantages and disadvantages, and also show how restrictor plates are used. Guidelines for the use of artificial cavities and restrictor plates are presented in 8C.

Artificial Cavities

Artificial cavities for red-cockaded woodpeckers were developed in the late 1980's and early 1990's (Copeyon 1990, Copeyon *et al.* 1991, Allen 1991, Taylor and Hooper 1991), and have since revolutionized management of red-cockaded woodpeckers. Prior to their development, biologists were unable to address the severe limitation in cavities impacting most populations, and therefore had little ability to slow, much less reverse, the decline of the species. With the advent of artificial cavity technology, cavities and entire clusters can be provided. In combination with aggressive habitat management, cavity management can stabilize and increase populations.

The power of the new technology to conserve and protect red-cockaded woodpeckers was illustrated soon after development, when Hurricane Hugo destroyed nearly 90 percent of the cavity trees on the Francis Marion National Forest in 1989. Rapid and extensive use of drilled cavities and cavity inserts following the devastation saved a large proportion of the population and allowed for population growth in subsequent years (Watson *et al.* 1995). During the 1990's, many other populations were

stabilized, and some increased, through cavity provisioning in combination with prescribed burning. In addition, other recently developed management tools such as translocation, mitigation, and Habitat Conservation Plans are based to a large degree on the use of artificial cavities.

However, artificial cavities have not always been used effectively. Widespread and haphazard installation of artificial cavities has negative impacts on red-cockaded woodpeckers and their potential cavity trees, and misdirects valuable management efforts and funds. Before artificial cavities are installed, managers should have a clear understanding of population dynamics in this species, especially the role of cavities and the effects of spatial structure on population growth or decline (see 2B, 2C). In addition, managers need to be well versed in the benefits and drawbacks of the various installation methods, so that they know what to expect of cavities already installed in their populations and can choose the appropriate method for additional cavities. Also, proper maintenance of artificial cavities is essential.

There are basically four methods of constructing artificial cavities: Copeyon-drilled cavities and starts, cavity inserts, and modified drilled cavities. Copeyon-drilled cavities and starts were developed at North Carolina State University (Copeyon 1990). Cavity inserts were developed at the Southeastern Forest Experiment Station of the U.S. Forest Service, Clemson University (Allen 1991). Taylor and Hooper (1991) created the modified version of Copeyon's drilled cavity.

Basically, drilled cavities are constructed by drilling two tunnels: first, an entrance tunnel which the birds will use, and second, an access tunnel which is then used by the drill operator to ream out the cavity chamber. The access tunnel is plugged and sealed after the chamber is constructed. The two drilled methods, Copeyon and modified drilled, differ in the dimensions of the access tunnel and consequently in their durability. Drilled starts are drilled entrance tunnels with a widened interior. Cavity inserts are pre-fabricated nestboxes inserted into an opening in the tree created with a chainsaw. More detailed descriptions of these techniques are given below, followed by a comparison of their relative merits and applications.

Construction of Copeyon-drilled Cavities and Starts

The Copeyon-drilled method of cavity construction is illustrated in Figures 1 and 2. Candidate trees for Copeyon-drilled cavities must have at least 15.2 cm (6 in) of heartwood and no more than 8.9 cm (3.5 in) of sapwood, and less sapwood is preferred.

To construct the cavity, a gasoline-powered drill equipped with a wood-boring bit 5.1 cm (2 in) in diameter is used to excavate an entrance tunnel through the sapwood and into the heartwood, at a slightly upward angle. The same bit is used to begin a second tunnel 5.1 to 10.2 cm (2 to 4 in) above the entrance tunnel. This access tunnel is then continued at a downward angle of roughly 60 degrees, using a 4.2 cm (1.65 in) bit, until the back of the entrance tunnel is intersected and 7.5 to 10 cm (3 to 4 in) below the entrance tunnel have been opened to form a rudimentary chamber. The rudimentary

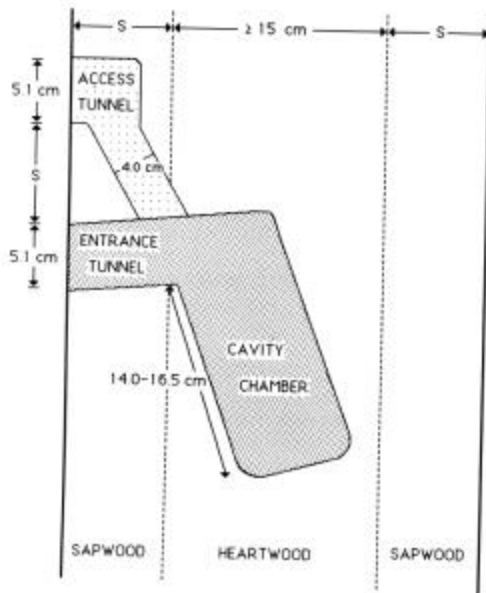


FIGURE 1. Diagram of Copeyon-drilled cavity (Copeyon 1990).
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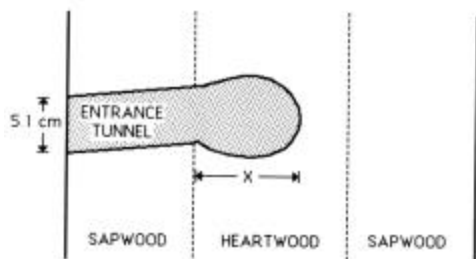


FIGURE 2. Diagram of Copeyon-drilled start (Copeyon 1990).
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chamber is then hollowed out, using the 4.2 cm (1.65 in) bit, to complete the cavity. The extent to which a cavity approaches the shape and dimensions of a naturally excavated cavity depends on the width of sapwood, the diameter of the heartwood core, and the skill of the drill operator. Care must be taken to avoid drilling into the sapwood at the front of the cavity chamber, by drilling at too steep an angle, or at the rear of the cavity, by drilling too deep.

The access tunnel is sealed with wood plugs and non-toxic wood putty. A thin, flexible wood veneer called “wiggle board” may be used to line the entrance tunnel instead of wood putty. A comprehensive maintenance schedule is required in the weeks immediately following construction, to inspect for resin leakage.

Upon completion of the cavity, resin wells are drilled with a 1.3 cm (0.5 in) twist bit or cut with a knife, and the area several feet above and below the cavity is scraped with a bark knife or hoe blade to give the tree the reddish appearance of an active red-cockaded woodpecker cavity tree. Non-toxic white paint is sprayed below resin wells, above and below the cavity entrance, and completely around the tree bole in the vicinity of the cavity to simulate natural pine resin.

Drilled starts are constructed using the above method to create an entrance tunnel (Figure 2). The access tunnel and cavity chamber are not constructed. Instead, a 4.2 cm (1.65 in) bit is used to enlarge the rear of the entrance tunnel (within the heartwood) to give the red-cockaded woodpecker room to excavate the cavity chamber. Such an advanced start may be large enough for a red-cockaded woodpecker to roost within, and red-cockaded woodpeckers can complete a drilled start in several months to a year (J. H. Carter III *et al.*, unpublished). Drilled starts can be placed in trees with too much sapwood and/or too little heartwood to accept a drilled cavity.

Construction of Modified Drilled Cavities

Taylor and Hooper’s (1991) modification of Copeyon’s drilled cavity technique differs from the original technique in that larger bits are used to begin the access tunnel (8.9 cm [3.5 in] bit) and to construct the vertical access tunnel and cavity chamber (7.6 cm [3 in] bit). Using this technique, most of the access tunnel and cavity chamber can be excavated at once. Resin wells are created and the trunk is painted to resemble a natural cavity tree just as described above.

Construction of Cavity Inserts

The cavity insert (Allen 1991) is a completely different approach to cavity construction. In this method, a chainsaw is used to carve a rectangular opening in a pine tree, and prefabricated wooden box containing a cavity is inserted into the opening. The cavity insert is superficially similar to a bluebird box and is secured in the excavated opening with wooden wedges and non-toxic wood putty. A full frontal restrictor plate is generally required to reduce or prevent damage by pileated woodpeckers. Because inserts are often placed in trees that are mostly sapwood, the box must be coated with a

waterproof epoxy to prevent resin leakage into the cavity chamber. Cavity inserts are held primarily within the sapwood of the tree, and so can be placed in pines that have little heartwood. Trees of at least 38.1 cm (15 in) in diameter at cavity height are required. Resin wells are created, and the trunk is painted, upon completion of the insert installation.

Comparison of Construction Methods

Preliminary work evaluating the four methods of cavity provisioning was conducted in the Francis Marion National Forest (Hooper and Taylor, unpublished), an appropriate location for such a study because of the large-scale provisioning of all cavity types following Hurricane Hugo. Although the population increased rapidly following the devastation of Hugo (Watson *et al.* 1995), a declining trend has been present since 1996 (Hooper and Taylor, unpublished; USFWS, unpublished). Aging of the artificial cavities is considered a potential contributing factor (in addition to problems implementing the prescribed burning program).

Hooper and Taylor's (unpublished) work suggests that Copeyon-drilled cavities and starts remain in use for a remarkably long period. After an average of 8.5 years, more than two-thirds of Copeyon-drilled cavities remained in use, and one quarter of the remaining available Copeyon-drilled cavities were in use as nest cavities. Half of all the original drilled starts were in use as cavities 8.5 years later, and one quarter of the remaining available cavities that were originally drilled starts were in use as nest cavities. Cavity inserts did not exhibit the same durability. Just less than half of cavity inserts remained in use after 8.5 years, and none were used as nest cavities. However, cavity inserts were constructed in sites of consistently lower quality than those in which drilled cavities were placed (D. Allen, pers. comm.). Because sites receiving inserts had suffered heavier hurricane damage and had virtually no old pines remaining after the storm, comparisons of durability between inserts and drilled cavities are biased by differential habitat quality. Modified drilled cavities showed the lowest durability of all cavity types, without the same systematic bias in habitat quality. Less than one third of modified drilled cavities were used an average of 7.3 years later, and none as nest cavities.

Differences in cavity survival did not appear to result from differential mortality of trees holding the various cavity types (Hooper and Taylor, unpublished). Less than 2 percent of pines with artificial cavities died from structural failure of the tree bole resulting from cavity installation, and this did not differ between trees containing inserts and those with drilled cavities. Cavity trees with inserts did not appear to suffer more damage from wind or physiological stress than other cavity trees, a conclusion also reached by Lowder (1995). Instead, lowered survival of inserts was due to higher rates of flooding and cavity enlargement. Inserts were not fitted with full restrictor plates (below), which would have reduced enlargement rates considerably. Almost half of all inserts had the interior altered by the birds to the point where the insert was breached and the tree itself was visible. Such expansion did not appear to affect the activity status of the inserts.

Lowered survival of modified drilled cavities was due to high rates of damage to the entrance tunnel and access plug. The larger access plug was far more likely to rot, and the septum between the access plug and entrance tunnel was more likely to be altered by decay or by other woodpeckers, than were those of Copeyon-drilled cavities. Enlargement of completed drilled starts was negligible.

Recommended Construction Methods

In light of the current value of cavity trees and potential cavity trees, we have formulated careful guidelines for the construction of artificial cavities (see 8C). Copeyon-drilled cavities are recommended for cavity provisioning if pines with sufficient heartwood are available. Managers may choose to drill starts instead of cavities if the cavities are not likely to be needed for a year or more. (Drilled starts over one year in age were found to be as useful to the birds as Copeyon-drilled cavities; Hooper and Taylor, unpublished.) Use of inserts is recommended when cavities are needed rapidly and there are no pines old enough to support a Copeyon-drilled cavity. Use of the modified drilled method of cavity construction is to be avoided.

When inserts are used, it is imperative that they are inspected carefully for possible damage, both before and after installation. To avoid flooding, the insert must be made so that the entrance tunnel enters the box at an upward angle, and there should be a hole, 0.95 cm (0.375 in) in diameter, drilled from the lower front of the box to the bottom of the cavity chamber, to allow water to escape. Also, all inserts must be fitted with full restrictor plates that are carefully maintained (see below), and all inserts must also must be coated with a thick layer of Acraglass gel, fiberglass, or acrylic resin prior to installation. A final concern in the use of inserts is the tendency of red-cockaded woodpeckers to breach the cavity chamber. This behavior has the potential to cause death of the birds from resin, although it is possible that excavation is slow enough that resin leaks are rare. Further study into this issue is necessary. Causes of the behavior are unknown, but may be related to the softness of the wood, western red cedar, often used in production. Use of harder wood for construction should be researched (but such research requires proper permits). Other measures that may reduce the likelihood of breaching include constructing inserts with somewhat thicker walls, and placing inserts only in trees large enough to support the rear third of the box firmly within heartwood. Several sizes of inserts can be used to best match trees of different sizes, although it is important to place inserts in the largest and oldest trees available. All inserts should be internally inspected at least once annually for resin leakage. If red-cockaded woodpeckers have excavated more than 1.3 cm (0.5 in) into the insert walls, the cavity should be plugged or replaced.

When Copeyon-drilled cavities and starts are used, it is imperative that they be screened and checked for resin leaks (below). All artificial cavities and starts must be inspected and maintained as described below and in section 8C.

Cavity Screening, Resin Leakage, and Maintenance Checks

All drilled starts and drilled cavities must be screened with heavy wire mesh (0.64 by 0.64 cm [0.25 by 0.25 in]) to prevent access by red-cockaded woodpeckers for at least four weeks after installation to ensure that no resin is leaking into the cavity chamber. If leaks are detected, cavities must remain screened and additional checks conducted. Persistent resin leakage into entrance tunnels can be treated using repeated scraping, applications of wood putty, replacement of wooden veneer, or redrilling with a 5.1 cm (2 in) diameter bit. If the leak is severe, cavities should be blocked with a wooden plug at least 7.6 cm (3 in) long and replaced elsewhere. Artificial cavities and starts should be constructed during the non-growing season (except in emergencies) to reduce the likelihood of resin leakage.

All artificial cavities, including inserts, and drilled starts should be checked for latent resin leakage during the first growing season after installation. If this check is negative no further maintenance checks are required for drilled starts and cavities unless the entrance tunnel begins to heal over from lack of red-cockaded woodpecker use. If an entrance tunnel is redrilled or scraped, screen it again as described above. Inactive artificial starts and cavities require periodically redressing of resin wells and the rescraping of bark to enhance the likelihood of discovery and occupation by red-cockaded woodpeckers. All inserts should be internally inspected for resin leaks once each year.

Cavity Height, Orientation, and Location

In general, artificial cavities should be placed as high as the recipient trees will allow, within the range of natural cavity heights in the surrounding habitat. Height of drilled cavities may be limited by the amount of heartwood present, and height of inserts may be limited by tree diameter; both will vary according to local conditions. For example, sites with low site index such as sandhills will support only low cavities. Cavities should be oriented so that the entrance faces west, because natural cavities show a tendency to be oriented in this direction (Locke and Conner 1983). Cavities should be constructed within 66 m (200 ft) of existing cavity trees to maintain the integrity of the cluster.

Number and Definition of Suitable Cavities

Carrie *et al.* (1998) found that group size of red-cockaded woodpeckers in Louisiana increased with the number of cavities provisioned, and recommended a minimum of three to four suitable cavities per cluster. Results of the study more clearly supported the use of four suitable cavities rather than three as a minimum. A minimum of four suitable cavities per cluster has also been the traditional policy of the U.S. Fish and Wildlife Service. We therefore recommend that each cluster contain at least four suitable cavities.

A suitable cavity has a single entrance, an entrance tunnel that is not enlarged, a cavity chamber that is not enlarged, a solid base, and is free of debris. In addition, the cavity plate must not contain large amounts of dead wood (Carrie *et al.* 1998). Relict, enlarged, or any suspect cavities must not be considered suitable for use by red-cockaded woodpeckers.

Restrictor Plates

The cavity restrictor was developed at North Carolina State University in the mid-1980's (Carter *et al.* 1989), to prevent and repair the enlargement of red-cockaded woodpecker cavity entrances. Cavity restrictors are square or rectangular metal plates with an inverted U-shaped or circular opening, 3.8 cm (1.5 in) wide, in the center of the plate. Typically, they are made of approximately 22 gauge stainless steel, and range in size from 7.6 by 7.6 cm (3 by 3 in) to much larger. Smaller restrictors are used for starts and cavity entrances that show little damage, while the largest sizes are used for enlarged cavities and to cover the front of cavity inserts.

The inverted U-shape opening was the original design (Carter *et al.* 1989). The opening extends from the entrance hole to the bottom of the restrictor plate, allowing the birds' feet to contact the tree surface when entering and exiting the cavity. If restrictor plates with circular openings are used, the metal directly below the opening of the entrance tunnel must be removed to allow the birds a secure foothold. Care must be taken to ensure that this metal is not so rough or jagged as to cause injury to the birds' toes or feet. Smooth, slick metal below the entrance is a deterrent to red-cockaded woodpecker use and may completely prevent use of some cavities.

Restrictors are attached to the tree with nails or screws at all four corners placed in pre-bored holes. Wood screws (1.3 cm [0.5 in] long) are preferred over nails because they allow easy repositioning of the restrictor with minimal damage. Screws or nails longer than 2.54 cm (1 in) should not be used because the cavity chamber may be breached, creating a hazard for cavity occupants. Restrictors should be painted brown with a non-toxic spray paint in order to blend with the tree and enhance acceptance by red-cockaded woodpeckers.

The primary use of restrictors is to repair or prevent enlargement of cavity entrances (see also 2F), usually done by pileated woodpeckers but occasionally by red-bellied and red-headed woodpeckers, northern flickers (*Colaptes auratus*), and gray squirrels (*Sciurus carolinensis*). Pileated woodpeckers can seriously damage cavities in just minutes, and can completely destroy cavities in less than an hour, but the reasons for this behavior remain unknown. Further, pileated woodpeckers may damage some cavities in a cluster, while leaving others unharmed. Some cavities, or entire clusters, can exist undamaged for years in areas frequented by pileated woodpeckers, then suffer a sudden onset of damage. In extreme circumstances, pileated woodpeckers can damage or destroy most or all cavities in a cluster, leading to cluster abandonment. Commonly, a cluster suffers chronic damage over several years, leading to cluster instability and eventual abandonment. Because of the critical importance of suitable cavities to red-

cockaded woodpeckers, use of restrictors to prevent and repair damage is an essential element of management for many populations. The number of cavities restricted in a cluster will vary according to circumstances, and may range from none to all cavities present. Knowing when to use restrictors to prevent damage, and when their use is not necessary, is a skill gained from experience and good judgment.

Whereas pileated woodpeckers can destroy red-cockaded woodpecker cavities by doubling the diameter of the entrance tunnel and exposing the cavity chamber, red-bellied woodpeckers, red-headed woodpeckers, and flickers normally enlarge cavity entrance tunnels and cavity chambers only enough to allow access. Over several years, these species can modify a cavity so that red-cockaded woodpeckers will rarely, if ever, use it. Although some rate of loss of red-cockaded woodpecker cavities due to modification by other species is natural, red-cockaded woodpeckers cannot always tolerate such losses in today's forests. In small, declining, or isolated populations, any loss of suitable cavities may not be tolerable. It will usually be necessary to use restrictors to repair enlargement by these species in such populations.

In the past, restrictors were sometimes used to exclude some avian cavity kleptoparasites, such as red-bellied woodpeckers, red-headed woodpeckers, and European starlings (*Sturnus vulgaris*), from cavities with either enlarged or unenlarged entrance tunnels. Variation in diameter of natural entrance tunnels allows access of some individuals or species to some cavities. For instance, both male and female red-bellied woodpeckers can enter some natural, unenlarged entrance tunnels, while only the slightly smaller females can access others. Eastern bluebirds and southern flying squirrels can access all cavities. However, use of restrictors on unenlarged cavities to exclude cavity kleptoparasites is not recommended, because of danger to red-cockaded woodpeckers. The difference between excluding a starling and excluding or entrapping a red-cockaded woodpecker is a matter of millimeters. Several deaths of adult red-cockaded woodpeckers resulting from entrapment in restricted cavities have been documented in the North Carolina Sandhills (J. H. Carter III *et al.*, unpublished). In most cases, the affected red-cockaded woodpecker had successfully entered the cavity, but could not exit. Given that population-level benefits of cavity kleptoparasitism have not been demonstrated (Kappes 1993, Conner *et al.* 1996, Mitchell *et al.* 1999; see 2F), there is little justification for use of restrictors to exclude kleptoparasites.

Restrictors must be inspected annually, because restrictors that have loosened or come out of place are a serious hazard to red-cockaded woodpeckers. In populations where annual monitoring is not likely, restrictors should not be used. Restrictors may have subtle costs as well: examination of a limited number of adult red-cockaded woodpeckers using restricted cavities showed visual evidence of the blunting of the bill (J. H. Carter III *et al.*, unpublished). Raulston *et al.* (1996) concluded that restrictors did not affect woodpecker survival or bill wear, but this was a small, short study and further research is warranted.

In summary, restrictors are an important management tool, but they must be used in the appropriate situations only, installed by experienced personnel, and monitored

annually. Widespread use of restrictors without specific need for them is not recommended, because they are potentially dangerous. Cavity restrictors are best used to prevent or repair enlargement of cavities by pileated woodpeckers. In small populations, their use against cavity damage by other species may also be necessary. Restrictors should not be used to prevent starlings and other woodpeckers from using the cavity, because red-cockaded woodpeckers can be entrapped as well.

C. PREDATOR AND CAVITY KLEPTOPARASITE CONTROL

Red-cockaded woodpecker populations that are healthy and of medium to large size require no predator control and few measures to combat cavity kleptoparasites. Predators and cavity kleptoparasites were not among the original causes of the decline of red-cockaded woodpeckers, and their removal or control will not result in population or species recovery. Critically small populations, however, may not be able to tolerate even occasional loss of nests or cavities. Therefore, managers of critically small populations (less than 30 potential breeding groups) may choose to use exclusion devices or other methods of control, but only in concert with aggressive management of foraging and nesting habitat. Methods of predator and kleptoparasite control are described in this section, and guidelines for their use are presented in 8E. A general discussion of predation, cavity kleptoparasitism, and cavity enlargement is given in 2F, and use of restrictors to control cavity enlargement is described in 3B and 8C.

Most control measures used in red-cockaded woodpecker populations have been designed for one of two taxa: flying squirrels and rat snakes. Methods vary from lethal measures to non-invasive techniques such as bark shaving (Saenz *et al.* 1999) and provision of nest boxes (Loeb and Hooper 1997). In general, the least invasive techniques are preferred.

Exclusion of Rat Snakes

Three artificial methods of excluding rat snakes from cavity trees have been explored: snake nets, snake excluder devices (SNED's), and the bark-shaving technique. Snake nets were developed by Neal *et al.* (1993b, 1998), and consist of a folded nylon monofilament net stapled to cavity trees at roughly 1.5 m (5 ft) above the ground. Rat snakes attempting to climb cavity trees get entrapped in the nets and soon die from heat stress. Red-cockaded woodpeckers can also get caught in these nets. Samano *et al.* (1998) reported the death of four red-cockaded woodpeckers and the entrapment of a fifth (rescued by biologists) in snake nets in a single year. Because of the documented danger to red-cockaded woodpeckers, use of snake nets is prohibited.

Snake excluder devices (SNED's) were developed by Withgott *et al.* (1995), and consist of a strip of lightweight aluminum flashing attached to the trunk of the cavity tree at ground level or up to 1.5 m (5 ft) above the ground. Withgott *et al.* (1995) used a 60 cm (23.6 in) wide band of aluminum flashing that they wrapped around and stapled to the bole of cavity trees. Prior to stapling the flashing in place, the bark on the bole of the

cavity tree was scraped to smooth the surface and permit a tighter fit. The bark was also scraped relatively smooth about 30 cm (1 ft) above and below each SNED after installation. SNED's proved to be highly effective in preventing climbing by rat snakes, and did not appear to affect use of the tree by red-cockaded woodpeckers (Withgott *et al.* 1995). Neal *et al.* (1998) reported numerous over-climbs of SNED's on red-cockaded woodpecker cavity trees in Arkansas and Mississippi that were fitted with narrow metal flashing (less than 0.9 m [3 ft]), whereas only one over-climb occurred on 92 cavity trees fitted with metal flashing greater than 0.9 m (3 ft) wide. Thus, SNED's greater than 0.9 m (3 ft) wide appear to be an effective, non-lethal method to reduce rat snake predation on red-cockaded woodpecker nest cavities. SNED's require adequate annual maintenance, to check for dangerous tears in the aluminum and to remove any resin accumulation.

Bark-shaving was recently developed by Saenz *et al.* 1999 as an effective means of deterring climbing by rat snakes. This modification of Withgott *et al.*'s (1995) technique is safer, less expensive, and aesthetically more pleasing than use of aluminum flashing (i.e., SNED's). A very sharp draw knife is used to shave the bark around the circumference of the tree in a 1 m (3.3 ft) band, at breast height, to eliminate furrows and rough surfaces without cutting into the cambium (Saenz *et al.* 1999). Breast height was chosen for ease of execution. This technique proved to be nearly 100 percent effective in experimental trials, and the one over-climb event occurred 3 ½ months after shaving on a tree that had developed a rough surface again (Saenz *et al.* 1999). Reshaping prevented the snake from climbing this tree again. Thus, bark-shaving is a promising new technique that can be used at the start of the nesting season or upon installation of artificial cavities, to give roughly three months of additional protection. Care must be taken not to damage the cavity tree by cutting into xylem tissue. Also, resistance to fire may be decreased by bark-shaving (Saenz *et al.* 1999), and any cavity tree thus treated should be well protected against fire.

The resin barrier created by red-cockaded woodpeckers is an extremely effective means of excluding rat snakes from cavity trees, especially in highly resinous longleaf pines (Ligon 1970, Dennis 1971b, Jackson 1974, 1978a, Rudolph *et al.* 1990a). In longleaf pine habitats, no additional measures are needed to control rat snakes regardless of population size. For critically small populations (less than 30 potential breeding groups) in pine types other than longleaf, managers may choose to install snake excluder devices or use the bark-shaving technique on trees likely to be used as nest trees. Managers may also choose to use bark-shaving to provide short-term protection against snakes when installing artificial cavities in any population, in any habitat type. Bark-shaving may be especially useful just before the nesting season, to protect active artificial cavity trees that do not yet have a resin barrier.

Exclusion of Southern Flying Squirrels

Southern flying squirrel excluder devices (SQED's) were developed by Montague *et al.* (1995), and consist of sheets of aluminum flashing that are wrapped around the cavity tree above and below the cavity entrance. Small portions of the flashing extend

perpendicular to the bole of the pine tree. If kept clean of hardened pine resin, the SQED's serve as an effective barrier and deny squirrel access to red-cockaded woodpecker cavities when they climb up and down the bole of cavity trees (Montague *et al.* 1995, Loeb 1996). However, a "skilled" flying squirrel can fly directly to a cavity entrance if adjacent pines are sufficiently close to permit a glide path. SQED's require inspection and maintenance at least yearly, to ensure no dangerous tears develop and to keep them free from resin. Again, SQED's are effective but their use is not necessary in populations of 30 or more potential breeding groups.

Montague *et al.* (1995) recommended that cavities reclaimed from southern flying squirrels be vacuumed to remove chewed pine needles and squirrel feces that are typically present in cavities with squirrels. Cavity cleaning may increase the probability that red-cockaded woodpeckers will reoccupy the cavity.

Lethal vs. Non-lethal Methods of Control

Rat snakes, southern flying squirrels, and other predators and kleptoparasites are all important components of southern pine ecosystems. Measures to control these species should not be applied in all areas managed for red-cockaded woodpeckers. Large and medium-sized populations located in areas of quality habitat should have sufficient reproduction and population size to easily offset any losses caused by predation and kleptoparasitism. However, in critically small populations (less than 30 potential breeding groups) where appropriate habitat is in the process of being restored, or where populations are being reintroduced, some level of predator and kleptoparasite management may be necessary. Provisioning more than four suitable cavities through the use of restrictors or artificial cavities would also potentially reduce the impact of competitors in small populations, but artificial cavities should not be installed wholesale throughout the landscape. The availability of non-lethal technology such as SNED's, bark-shaving, and SQED's makes it possible to deter predators and competitors without negatively impacting other biological components of the forest ecosystem. Also, the retention and creation of adequate numbers of snags and/or nestboxes for potential kleptoparasites (Harlow and Lennartz 1983, DeFazio *et al.* 1987, Kappes and Harris 1995, Loeb and Hooper 1997) is an important management option. Use of lethal devices and euthanasia to control predators and kleptoparasites is discouraged.

D. TRANSLOCATION

Translocation is the artificial movement of wild organisms between or within populations to achieve management objectives. It is an important tool for the management and recovery of red-cockaded woodpeckers, if used in the appropriate situations and in the appropriate manner. In this section, we describe the reasons for using translocation and give a brief review of its use and success in red-cockaded woodpecker management. Guidelines for its use are presented in 8F.

Translocation of red-cockaded woodpeckers has three specific applications for which it is best suited: (1) augmentation of a population in immediate danger of extirpation, (2) development of a better spatial arrangement of groups, to reduce isolation of groups or subpopulations, and (3) management of genetic resources. We refer to the first application as population augmentation. This consists of moving birds from a healthy donor population to a critically small recipient population (less than 30 potential breeding groups). We refer to the second application as strategic recruitment, which is achieved by moving birds from within or between populations to recruitment clusters strategically located to link groups and subpopulations. All translocations, including those intended to augment a population, should serve to develop better spatial arrangements of groups.

Population augmentation is a means of buffering at-risk recipient populations against effects of demographic and environmental stochasticity (see 2C), which can result in extirpation critically small populations regardless of other management efforts. This management action also serves to counteract the inbreeding depression that can reduce the persistence of very small, isolated populations (Haig *et al.* 1993, Daniels *et al.* in press). Augmentation is not necessary for larger populations because they are not so highly vulnerable to stochastic events (other than catastrophes).

Strategic recruitment is a means to develop the beneficial spatial arrangements that can dramatically increase persistence and health of red-cockaded woodpecker populations (Conner and Rudolph 1991b, Crowder *et al.* 1998, Letcher *et al.* 1998). Linking isolated groups and subpopulations with newly established breeding groups may be a slow process, because each new cluster must be within helper dispersal distance of active clusters. However, over time strategic recruitment can optimize spatial arrangements of groups within populations.

For the purposes of augmentation or strategic recruitment, a potential mate can be moved to a cluster inhabited by a solitary individual (mate provisioning), or potential pairs can be moved in tandem to unoccupied clusters. Currently, translocations are always performed in a way that links a translocated bird to a target cluster. Future translocations may involve releases of birds not necessarily linked to a specific target cluster but rather a group of clusters. Two other applications of translocation include its use during mitigation (see 4A) and the establishment of new populations in formally occupied and restored habitat (reintroduction). An application for future use is the translocation of individuals among recovered populations and essential support populations to counteract species-wide genetic drift (see 2C).

Translocation has its benefits and drawbacks. It can be an important method to counteract loss of genetic variation but may also serve to disrupt valuable local genetic resources. It is an especially useful tool in the management of red-cockaded woodpeckers, because population dynamics in this species are regulated by the number of potential breeding groups in a population, not the annual number of young produced (Walters 1991; see 2B). Therefore, some juvenile birds may be moved without affecting the overall population size or trend. However, impacts to the donor areas and populations

must be carefully evaluated and controlled (Griffith *et al.* 1989, Haig *et al.* 1993). Moreover, translocation must not be used as a substitute for habitat management and restoration, two more difficult but much more important management tasks (e.g., Pitelka 1981, Meffe 1992). Causes of population decline should always be identified and removed before translocation is attempted (Short *et al.* 1992, Meffe 1992, Caughley 1994). Thus, in general, translocation of red-cockaded woodpeckers is a short-term tool to be used in specific crisis situations with utmost caution and only after habitat suitable in quality and quantity exists (Griffith *et al.* 1989, Kleiman 1989) and habitat management plans emphasizing frequent fire are fully implemented. In addition, translocation may have a long-term application among recovered populations to counteract species-wide genetic drift, if natural dispersal is deemed insufficient for adequate gene flow.

History of Translocation of Red-cockaded Woodpeckers

Prior to the development of artificial cavities (Copeyon 1990, Allen 1991) and translocation (DeFazio *et al.* 1987), many managers and biologists were pessimistic about the long-term persistence of red-cockaded woodpeckers (Ligon *et al.* 1986, Escano 1995). In particular, there was little hope of conserving and restoring the many small, declining populations. Recently, however, most populations have been stabilized and/or increased (Hooper *et al.* 1990, Richardson and Stockie 1995, Watson *et al.* 1995, Walters and Meekins 1997, Walters *et al.* 1997). For some small populations, increases in population size were achieved through aggressive habitat management and cavity provisioning without resorting to translocation (Richardson and Stockie 1995, Watson *et al.* 1995, USFWS unpublished, Walters and Meekins 1997, Walters *et al.* 1997). However, the stabilization and increase of other critically small populations has required the use of translocation in concert with intensive habitat and cavity management (DeFazio *et al.* 1987, Allen *et al.* 1993, USFWS unpublished).

Initially, translocations were performed as emergency efforts to rescue individual birds from military construction impacts (e.g., Odom *et al.* 1982) or loss of habitat to timber harvests (e.g., Reinman 1984). These early efforts met with very little success, and several authors criticized the use of translocation especially as mitigation for destruction of occupied clusters (Cely 1983, Jackson *et al.* 1983). Odom (1983) concluded, “red-cockaded woodpecker relocation is not recommended as a management tool at this time”, but also noted its potential and called for further research. Following these initial attempts in the early 1980's, experiments were performed in the late 1980's and early 1990's to test translocation methods and its usefulness as a recovery tool (Allen *et al.* 1993, Costa and Kennedy 1994).

Perhaps the best known of these experiments in translocation was the extremely intensive effort to conserve and restore the critically endangered red-cockaded woodpecker population in the Savannah River Site in South Carolina (Allen *et al.* 1993, Gaines *et al.* 1995, Franzreb 1999). By late 1985, this population was reduced to one breeding pair and two solitary males (DeFazio *et al.* 1987) and aggressive management was begun, including habitat management, cavity installation, and translocation (Gaines

et al. 1995). From 1986 to 1995, 54 red-cockaded woodpeckers were translocated, including 21 translocated from four donor populations outside the study area and 33 from within the population (Franzreb 1999). By 2000, the Savannah River Site population consisted of 31 potential breeding groups (P. Johnston, unpublished). Clearly, translocation was an important part of the dramatic change in this population's status.

Following the success of the Savannah River Site translocation attempts (Allen *et al.* 1993), the Southern Region of the U.S. Forest Service decided to implement red-cockaded woodpecker translocations as a management tool in 1989 (Escano 1988). Because the Apalachicola National Forest in Florida contained the largest and only recovered red-cockaded woodpecker population, it was chosen as the primary donor population. From 1989 to 1992, 18 red-cockaded woodpeckers were translocated from the Apalachicola NF to seven other national forest units (Hess and Costa 1995). Currently, translocation remains an important crisis management tool to be used with caution in appropriate circumstances.

Translocation Success

Efforts to measure the success of translocation as a management technique have been hampered by inconsistent data collection and differing definitions of success (Costa and Kennedy 1994). Definitions of success have varied, ranging from the individual being present soon after release to the fledging of offspring the following breeding season (Costa and Kennedy 1994). To further confuse the issue, definitions of success must change depending upon the objective of the translocation: for augmentation of a critically small population, reproduction of a translocated bird anywhere in the population would be considered successful; however, if the objective is strategic recruitment of a new group by translocating birds from within the population, then reproduction of those individuals in clusters other than the target would not be considered a success.

Currently, the average estimated success rate for translocation is roughly 50 to 60 percent, for various meaningful definitions of success including presence in the recipient cluster in the following breeding season (Hess and Costa 1995), evidence of breeding in the following season or of pair-bonding just prior to the breeding season (Costa and Kennedy 1994), and remaining at or near the release site for 30 days (Franzreb 1999). Similarly, Franzreb (1999) reported that roughly half of adults and subadults (25 of 49) translocated to and within the Savannah River Site reproduced somewhere within that population. Higher success has been reported for simultaneous movement of multiple pairs (50 to over 70 percent present in the following breeding season; Carrie *et al.* 1999, USFWS unpublished), an encouraging development in translocation methods for red-cockaded woodpeckers and one which has been emphasized for other species as well (Griffith *et al.* 1989). Reproduction specifically at the recipient cluster is currently estimated to have occurred in 27 percent of translocations conducted between 1989 to 1995 (48 of 178, Edwards and Costa, in prep.).

Success of translocations has increased as methods have improved. Information is slowly accumulating on the effects of age, sex, and other factors such as distance,

habitat condition, and the number of birds released on the likelihood of successful translocation. This research has been invaluable in formulating both a regional translocation strategy and specific guidelines for the movement of birds. Researchers agree that moving females to territories with solitary males, and moving potential pairs simultaneously, are the most successful types of movements (Rudolph *et al.* 1992, Allen *et al.* 1993, Costa and Kennedy 1994, Hess and Costa 1995, Edwards and Costa, in prep.). Birds are less likely to return to their original cluster if moved more than roughly 19.3 km (12 mi; Allen *et al.* 1993, Franzreb 1999). Other factors, such as insufficient number or poor condition of recipient cavities, problems in transport, and problems at the time of release, reduce success of translocations (Hess and Costa 1995). Finally, Rudolph *et al.* (1992) suggested that simultaneous movement of multiple pairs (5-10) might increase success. Again, this method has yielded encouraging results: Carrie *et al.* (1999) reported a success rate, defined as birds present in the following breeding season, of over 70 percent (12 of 17) after releasing multiple pairs in the Sabine National Forest, and other translocations of multiple pairs have shown success rates from 50 to over 70 percent as well (USFWS unpublished).

In summary, it is apparent that translocation has an important but very specific role in the conservation and recovery of red-cockaded woodpeckers. It is not to be used as a substitute for more fundamental management actions that provide good quality foraging and nesting habitat. In the presence of good quality foraging and nesting habitat, translocation can be an effective short-term tool to counteract effects of demographic and environmental stochasticity and a useful measure over the long-term to reduce loss of genetic variation in isolated populations. Translocation is best performed by moving multiple pairs of red-cockaded woodpeckers simultaneously, to recruitment clusters that are strategically located to improve the spatial structure of the population.

E. SILVICULTURE

Silviculture is the theory and practice of controlling the establishment, composition, structure, and growth of forests to achieve management objectives (Smith 1986). It was developed primarily for the purpose of timber production, but can be used for other purposes including biological conservation (Smith 1986, Thompson *et al.* 1995). Silviculture is an important tool for the management of red-cockaded woodpeckers with or without the additional goal of timber production. Today's forests differ substantially in structure and species composition from the precolonial forests that supported red-cockaded woodpeckers in abundance (Conner and Rudolph 1989, Foti and Glenn 1991, Ware *et al.* 1993, Masters *et al.* 1995, Noel *et al.* 1998). Second growth forests are dense, contain many small young trees and few large old trees, and often have a complex vertical structure. Proper silviculture can restore and maintain the open, two-layered habitat required by red-cockaded woodpeckers. In this section, we discuss the compatibility and usefulness of silvicultural methods to management and recovery of red-cockaded woodpeckers. We give guidelines for the use of silviculture in 8H.

Conservation and recovery of red-cockaded woodpeckers are compatible with timber production within certain constraints (Rudolph and Conner 1996, Engstrom *et al.* 1996, James *et al.* 1997, in press, Hedrick *et al.* 1998). Suitable forest structure and function must be retained to support red-cockaded woodpecker populations. Suitable forest structure includes a substantial amount of large pines, low densities of small and medium sized pines, sparse or absent hardwood midstory, and abundant diverse herbaceous groundcovers (Hardesty *et al.* 1997, James *et al.* 1997, in press, Hedrick *et al.* 1998, Walters *et al.* 2000). Foremost among important functions of southern pine forests is the ability to carry frequent growing season fires (Platt *et al.* 1988b, Engstrom *et al.* 1996).

Silvicultural methods can be divided into three systems: even-aged, two-aged, and uneven-aged management. (Two-aged is sometimes included within even-aged management). Each system has several possible methods of regeneration, which is the simultaneous harvest and establishment of reproduction of trees (Thompson *et al.* 1995). Even-aged management includes clearcutting, standard seed tree, and standard shelterwood methods. Two-aged management includes modified seed tree and irregular shelterwood methods, and uneven-aged management includes single tree selection and group selection methods. Several researchers have assessed the compatibility of these methods with restoration and maintenance of habitat for red-cockaded woodpeckers (USFWS 1985, Lennartz 1988, Walker and Escano 1992, Walker 1995, USFS 1995, Rudolph and Conner 1996, Engstrom *et al.* 1996, Hedrick *et al.* 1998). The suitability of each method varies with forest type, silvicultural history, ownership, and management objectives.

Silvicultural systems also differ in methods of regulation, which is the control over the forest's ability to produce an even supply of timber over time (Baker *et al.* 1996). Regulation is an essential aspect of silviculture for sustained timber production but may not be relevant to biological conservation.

Silvicultural Systems

Even-aged Management

Even-aged management is the culture of trees of one age class in a given stand. The forest is regulated at the landscape level, with equal areas in each age class. Regeneration methods of even-aged management differ in the amount of residual trees remaining after harvest. Clearcutting is the removal of all commercially valuable trees on site. In standard seed tree and shelterwood methods, residual trees are left standing as seed sources after the initial harvest and are removed following the establishment of reproduction. Regardless of regeneration method, intermediate thinnings are made to improve growth and health of trees by reducing tree density (Smith 1986, Walker 1995). Modified seed tree and irregular shelterwood are not included as even-aged management in this document (see Two-aged Management below).

Clearcutting, standard seed tree, and standard shelterwood methods are not generally compatible with management to recover red-cockaded woodpeckers, except when used to restore native pines. Even-aged silviculture results in fragmented habitat, and red-cockaded woodpeckers are especially sensitive to negative impacts of habitat fragmentation because of their cooperative breeding system (see 2B). Even-aged silviculture renders stands unsuitable as nesting or foraging habitat for decades. Even with long rotations, even-aged silviculture results in the removal of the large old trees most important to red-cockaded woodpeckers. Even-aged silviculture can be useful in the removal of off-site pine species to restore native pines (see 3G). If within occupied habitat, such restoration is best limited to small areas (Ferral 1998).

Two-aged Management

Two-aged management is a modification of even-aged management in which two age classes exist in a given stand (Smith 1986, Rudolph and Conner 1996). Two-aged stands are created by modified seed tree and irregular shelterwood methods, which are similar to corresponding standard methods except that residual trees are never harvested. The forest is regulated in the same way as in even-aged management. Intermediate thinnings are important to reduce stand density.

Modified seed tree and irregular shelterwood methods are compatible with management of red-cockaded woodpeckers (Conner *et al.* 1991b, Rudolph and Conner 1996, Hedrick *et al.* 1998). Two-aged silviculture promotes the growth of old and even very old trees in every stand, and older trees are important to both nesting and foraging (see 2D, 2E). Prescribed burning can be conducted throughout much of the forest without fear of damaging young pines, because pine reproduction is concentrated in limited areas. This is a strong advantage in forests of loblolly and/or shortleaf pines which are sensitive to fire when young (Farrar 1996, Hedrick *et al.* 1998). Finally, two-aged silviculture can open up the forest and establish lower pine densities preferred by red-cockaded woodpeckers (Conner *et al.* 1991b). Irregular shelterwood and modified seed tree methods are the cornerstone of restoration of the shortleaf pine/bluestem grass (*Andropogon* and *Schizachyrium* spp.) ecosystem on the Ouachita National Forest in Arkansas (USFS 1996).

Modified seed tree and irregular shelterwood methods have some drawbacks in their application for red-cockaded woodpecker management. The older residual pines are subject to increased windthrow, especially the more shallow rooted pine species (Smith 1986), and increased lightning strikes. Reduction in canopy cover may reduce needle litter, an important fuel (Engstrom *et al.* 1996). An excessive pine midstory can develop, with detrimental effects on cluster occupancy (see 2D) and suitability of the stand for foraging (see 2E). However, suitably long rotations (see guidelines in 8H) can limit the areas with excessive pine midstory. Also, pine midstory may be more of a problem when dense stocking is desired for future timber harvests. For longleaf, shortleaf, and perhaps even loblolly sites, prescribed burning may effectively control midstory if the stand is to be kept at the lower pine densities preferred by red-cockaded woodpeckers.

Finally, modified seed tree and irregular shelterwood methods may not retain sufficient densities of large trees for newly regenerated stands to qualify as foraging habitat (see 8G). When using these methods in the presence of red-cockaded woodpeckers, long rotations are necessary to provide suitable foraging habitat.

Uneven-aged Management

Uneven-aged management results in stands with at least three age classes (Smith 1986). Reproduction occurs throughout the forest in gaps created by the harvest of single trees or groups of trees (regeneration by single tree and group selection, respectively). If group selection is used, patches of trees removed are generally below 0.8 ha (2 ac) in size. The forest is regulated at the stand level, usually by either timber volume or stand structure. The forest is regulated by timber volume using the volume/guiding diameter limit (V-GDL) method (Reynolds 1959, Baker *et al.* 1996, Farrar 1996, Guldin and Baker 1998), or by stand structure using the BDq method (Marquis 1978, Baker *et al.* 1996, Farrar 1996, Guldin and Baker 1998).

The V-GDL method uses periodic inventories to measure tree growth, which is then established as the allowable harvest. The guiding diameter limit is the size above which volume of trees meets the allowable cut. All trees above the guiding diameter limit are not necessarily cut; for every tree above the limit retained, an equal volume of trees below the limit are harvested (Farrar 1996, Guldin and Baker 1998). According to Guldin and Baker (1998), the classic marking rule for this method is to “cut the worst trees and leave the best”. In general, the V-GDL method of regulation is somewhat subjective and therefore can be difficult to apply (Farrar 1996, Guldin and Baker 1998).

The BDq method uses three parameters to describe the target after-cut stand structure: residual basal area (B), maximum diameter retained (D), and the ratio of number of stems in a given size class to those in the next larger class (q). The priority of these parameters is in the order given, so that trees above the maximum diameter are retained if residual basal area cannot be met without them (Baker *et al.* 1996, Farrar 1996, Guldin and Baker 1998). If the structure of the residual stand closely corresponds to q, the stand has a negative exponential (inverse-J) size distribution and is said to be well-balanced (Guldin and Baker 1998). Both q and D can be adjusted to increase the presence of large old trees to meet management objectives (Farrar 1996). The BDq method is preferred over the GDL method for most uses because it provides an objective means of monitoring the smaller size classes (Farrar 1996, Guldin and Baker 1998).

Uneven-aged management is compatible with restoration and maintenance of red-cockaded woodpecker habitat (Engstrom *et al.* 1996, James *et al.*, in press). Uneven-aged management can provide large old trees throughout the landscape. Densities of small and medium sized pines can be controlled to avoid detrimental effects on red-cockaded woodpeckers. Frequent prescribed burns can be used to control hardwoods and maintain herbaceous groundcovers in longleaf forest types. For loblolly and shortleaf forests, it is harder to use prescribed fire in uneven-aged stands because of fire sensitivity of young pines and the presence of young pines throughout the landscape (Rudolph and

Conner 1996, Hedrick *et al.* 1998). However, prescribed burning at intervals of variable length may be used successfully in these forest types (Cain 1993, Farrar 1996, 1998, Cain *et al.* 1998). Annual and biennial fires interspersed with periods of up to 5 years without fire may effectively control midstory and encourage herbaceous groundcovers while allowing for reproduction of loblolly and shortleaf pines (Cain 1993, Cain *et al.* 1998). The Red Hills region of south Georgia and north Florida supports a large population of red-cockaded woodpeckers in longleaf systems effectively managed with a combination of single tree and group selection methods (Engstrom and Baker 1995, Engstrom *et al.* 1996). Finally, uneven-aged management has been used successfully to remove off-site pine species and restore native pines (e.g. McWhorter 1996).

There are several drawbacks in the application of uneven-aged silviculture to the management of red-cockaded woodpeckers. The number of harvests, and consequently ecosystem disturbance, can be greater than that of two-aged management (Rudolph and Conner 1996) although this is not necessarily so (Engstrom *et al.* 1996, Farrar 1996). Application of prescribed fire is difficult or at least somewhat complex in uneven-aged stands of loblolly and shortleaf pines, and therefore hardwoods may become a problem (Rudolph and Conner 1996, Hedrick *et al.* 1998). Finally, selection systems, just like even-aged management, can result in the harvest of the old, large trees most valuable to red-cockaded woodpeckers. With careful application these drawbacks can be minimized.

Low Intensity Management

Some woodpecker populations exist in forests that are not managed for timber production. Low-intensity management for the primary purpose of biological conservation uses frequent growing season burns to control hardwoods, prepare the site for pine reproduction, and encourage beneficial native groundcovers. Natural disturbances such as wind-throw and lightning strikes establish gaps in the canopy for reproduction and recruitment to occur. Longleaf, shortleaf, and other pines on native sites are suited for low intensity management.

Some forests may require restoration prior to the application of this silvicultural method. Hardwood midstories and/or overstories may need removal. Herbaceous groundcovers may need to be restored, and dense pine stands will require thinning to densities suitable for red-cockaded woodpeckers (below).

Low intensity management is advantageous for red-cockaded woodpeckers because conservation is the primary goal. Low-intensity management offers aesthetic and recreational benefits as well, because the low tree density and healthy herbaceous layer are generally appealing to the public. Low-intensity management does not have the monetary benefits of timber production.

Pine Density

Pine densities generally recommended for timber production by uneven-aged management are 10.3 to 17.1 sq. m basal area per ha (45 to 75 sq. ft per acre) in longleaf

systems and somewhat higher for shortleaf and/or loblolly (Farrar 1996). Pine density before and after selection cutting generally remains within this range, which is just slightly higher than that recommended for red-cockaded woodpeckers. Even-aged and two-aged management typically result in pine densities of 18.3 to 27.4 sq. m basal area per ha (80 to 120 sq. ft per acre) or more (Farrar 1996), and after cutting densities are often reduced to below 2.6 sq. m per ha (20 sq. ft per acre). In addition, second-growth forests are generally more dense than old-growth woodlands (Ware *et al.* 1993, Masters *et al.* 1995, Noel *et al.* 1998). For management of red-cockaded woodpeckers in longleaf systems, pine densities should be between 40 and 60 sq. ft per acre. Two recent studies of foraging ecology documented increases in fitness of woodpeckers at these lower pine densities (Walters *et al.* 2000, James *et al.*, in press). James *et al.* (in press) recommends a target basal area of 11.2 sq. m per hectare (49 sq. ft per acre), and Walters *et al.* (2000) recommends basal areas between 10 and 14 sq. m per ha (40 and 60 sq. ft per acre). Further research is necessary to assess effects of pine densities on foraging ecology of woodpeckers in shortleaf and loblolly systems. However, pine densities below 18.4 sq. m per ha (80 sq. ft per acre), or an average spacing of at least 7.6 m (25 ft) between pines in mature stands, have been recommended to reduce risks of southern pine beetle infestations in shortleaf/loblolly forest types (Thatcher *et al.* 1980, Nebeker and Hodges 1985, Hicks *et al.* 1987, Belanger *et al.* 1988, Mitchell *et al.* 1991).

Priority for Leave Trees

Leave trees are those that remain standing after thinnings and harvests. Benefits to red-cockaded woodpeckers can be increased by preferentially leaving trees important to them. These important trees include older, old, and very old pines including relict and remnant pines, flat-tops, potential cavity trees (pines over 60 years in age), and pines scarred by turpentine harvest or lightning.

Site Preparation

Regardless of the silvicultural system used, some form of site preparation is necessary to establish pine reproduction. Site preparation removes vegetation and other organic material to expose the mineral soil required for seed germination. Prescribed burning is the best method of site preparation, because it mimics natural processes, minimizes disturbance to the soil, and promotes native herbaceous groundcovers beneficial to red-cockaded woodpeckers (see 2E). Prescribed burning during the growing season induces flowering of many native herbaceous plants (Platt *et al.* 1988a; see 2G).

Prescribed burning within one year of a good seed crop is generally the only site preparation needed, if hardwoods are well under control. If prescribed burning cannot be used, the Bracke scarifier-moulder or a roller drum chopper has fewer impacts than do disking, root raking, windrowing, and bedding. Any method of site preparation that disturbs the soil will favor ruderal, disturbance-tolerant grasses and forbs over desired species such as wiregrass (Provencher *et al.* 1998, 1999), and recovery of groundcovers can be exceedingly slow. For example, Provencher *et al.* (1997, 1998) found that

recovery of groundcovers following selective harvest of longleaf pine can take 50 years in deep sandy soils.

F. PRESCRIBED BURNING

Because of fundamental changes in the landscape and natural fire regime of the southeast, prescribed burning is and will continue to be the primary means of restoring and maintaining fire in southern pine ecosystems (Frost 1998). Prescribed burning is an essential management tool for the conservation and recovery of red-cockaded woodpeckers. Additionally, prescribed burning provides benefits for a suite of species characteristic of southern pine ecosystems (Robbins and Myers 1992, Costa 1995a).

Red-cockaded woodpeckers are rightly termed an umbrella or flagship species, because their protection and management provides for the conservation of entire ecosystems and the hosts of associated species within. It is especially prescribed burning, but also retention of old-growth and mature trees, that provides critical support for associated species. To maximize these benefits, the frequency, intensity, and season of prescribed fire should mimic the historic natural fire regime as closely as possible (Masters *et al.* 1996).

In this section, we briefly review the benefits of prescribed burning to red-cockaded woodpeckers and other species of southern pine ecosystems, and then address concerns about possible negative effects on some animals. We also review the application of prescribed fire to the landscape and its use in habitat restoration. A general discussion of the history and role of fire in southern pine ecosystems is given in 2G. Guidelines for the use of prescribed burning are given in 8I.

Benefits of Prescribed Burning

Benefits to Red-cockaded Woodpeckers

Red-cockaded woodpeckers require open woodlands for nesting and roosting cavities. Hardwood invasion eventually results in the abandonment of clusters and severe population decline (Beckett 1971, Hopkins and Lynn 1971, Van Balen and Doerr 1978, Locke *et al.* 1983, Hovis and Labisky 1985, Conner and Rudolph 1989, Loeb *et al.* 1992, Masters *et al.* 1995). Invasion of hardwoods and woody shrubs also degrades the quality of foraging habitat (James *et al.* 1997, Walters *et al.* 2000). Prescribed burning is a highly effective means of controlling such hardwood and shrub invasion. Prescribed burning can effectively control hardwoods and shrubs without damaging the herbaceous layer and soils. Prescribed fire also has direct benefits to herbaceous plants in southern pine communities by initiating flowering (Platt *et al.* 1988a). Fire helps maintain a healthy native plant community, which in turn leads to increased fitness of red-cockaded woodpeckers (Hardesty *et al.* 1997, James *et al.* 1997, in press). The mechanism for increased fitness of red-cockaded in the presence of abundant herbaceous groundcovers

has not yet been clearly demonstrated, but likely involves increased numbers and/or nutrient content of prey (James *et al.* 1997).

Benefits to Associated Species

Many plants and animals associated with southeastern pine communities are threatened by loss of habitat through fire suppression and conversion to other land uses. Management for red-cockaded woodpeckers directly supports these sensitive, threatened, and endangered species. Currently, over 120 species of plants and 56 animal species associated with red-cockaded woodpecker habitats are on the regional list of proposed, endangered, threatened, and sensitive species (USFS 1995). Many more herbaceous plants of longleaf communities are rare in today's landscape (Walker 1993), nearly all of which are adapted to growing season fire. Thirty-five percent of the amphibians and reptiles inhabiting longleaf pine forests, and 56 percent of the longleaf pine specialist species, were listed by at least one conservation agency as being of special concern (Guyer and Bailey 1993). Fire suppression was identified as a primary cause of the decline of these species. Fire benefits shortleaf pine communities as well, although these have not received as much research attention as longleaf systems. Masters *et al.* (1998) reported that species richness and diversity of small mammals increased in relation to midstory removal and prescribed fire, and no species was adversely affected by fire.

Prescribed burning directly benefits bird species associated with open pine woodlands such as Bachman's sparrows (*Aimophila aestivalis*), brown-headed nuthatches (*Sitta pusilla*), pine warblers (*Dendroica pinus*), prairie warblers (*D. discolor*), and red-headed woodpeckers (Engstrom *et al.* 1984, Jackson 1988, Wilson *et al.* 1995, Conner and Dickson 1997, J. Allen, in prep.). Bachman's sparrows, in particular, are in decline throughout most of their range and respond strongly to management for red-cockaded woodpeckers (Dunning and Watts 1990, Plentovich *et al.* 1998). Bird species associated with riparian habitats within open pine woodlands, such as Carolina wrens (*Thryothorus ludovicianus*), white-eyed vireos (*Vireo griseus*), common yellowthroats (*Geothlypis trichas*), and hooded warblers (*Wilsonia citrina*; Engstrom *et al.* 1984, Conner and Dickson 1997, J. Allen, in prep.), can benefit from prescribed burning as well. Riparian habitats within open pine forests, when frequently burned, support increased density and diversity of shrubs. The increased floristic and structural diversity is the likely cause of increased abundance of associated bird species (J. Allen, in prep.).

Concerns about Negative Effects

Increasing use of prescribed fire has prompted concern among some land managers, researchers, and the general public. A common anxiety is that prescribed burning during the growing-season may have detrimental effects on non-target species. Managers perceive negative impacts on game species, including losses of nests of ground-nesting birds such as northern bobwhites (*Colinus virginianus*) and wild turkeys (*Meleagris gallopavo*), and reduction of hard mast forage for game birds, white-tailed deer (*Odocoileus virginianus*), and black bear (*Ursus americanus*) among others. However, these concerns have not been substantiated. In fact, increases in abundance of

bobwhites and wild turkeys after the introduction of growing season burns have been reported in many areas (Landers *et al.* 1995, Palmer and Hurst 1998). Prescribed burning and pine thinning benefit white-tailed deer by increasing the production of available forage and preferred woody browse to more than four times that of untreated areas (Masters *et al.* 1996).

One immediate effect of growing season fire is the destruction of nests, and this has been a cause of concern for some people. However, for species associated with southeastern pine habitats, the benefits of prescribed burning far outweigh the occasional loss of nests. Improved habitat quality enables higher population densities, whereas fire suppression substantially lowers the abundance of these bird species (J. Allen, in prep.). Saving some nests through fire suppression can serve no purpose if the birds have no habitat in which to exist. In addition, many birds adapted to southeastern pine habitats, such as Bachman's sparrows, pine warblers, prairie warblers, and others, readily renest upon loss of a nest. Game birds such as wild turkeys and northern bobwhites also readily renest (Vangilder and Kurzejeski 1995, Harper and Exum 1999). This behavior acts to minimize any negative effect that fire can have.

There also has been some concern about possible effects of management for red-cockaded woodpeckers on neotropical-nearctic migratory birds. Some species of neotropical-nearctic migrants have experienced declines in recent decades (Robbins *et al.* 1989, Sauer and Droege 1992, Peterjohn and Sauer 1994). In response, conservation biologists and land managers have focused on these species. However, in the southeastern coastal plains, neotropical migrants of greatest management concern are largely associated with bottomland riparian forests (Hunter *et al.* 1994), whereas resident bird species of concern are associated with mature open pine forests and benefit from woodpecker management (Dunning and Watts 1990, Hunter *et al.* 1994, Wilson *et al.* 1995, Tucker *et al.* 1996). A study of the response of breeding bird communities to red-cockaded woodpecker management in southern Mississippi reported that 7 of the 9 bird species favored by woodpecker management were early successional or mature pine-grassland species under regional or national decline, whereas all 4 species favored by fire suppression were relatively common forest interior species exhibiting stable or increasing trends (Burger *et al.* 1998). In addition, almost all species of birds that increase abundance under fire suppression, such as red-eyed vireos (*V. olivaceus*), black-and-white warblers (*Mniotilta varia*), and Acadian flycatchers (*Empidonax virescens*), also use frequently burned riparian habitats within open pine ecosystems (J. Allen, in prep.).

Thus, management for red-cockaded woodpeckers benefits other resident bird species of concern without impacting those neotropical migrants that are in decline. Managers should not hesitate to conduct prescribed burns for fear of impacts to neotropical migratory birds. Neotropical-nearctic migrant species of concern will best be conserved not by fire suppression but by the protection of habitats most important to them, such as southeastern bottomland hardwoods and northeastern boreal forests.

Close proximity of human development to forests supporting red-cockaded woodpecker populations presents significant risks of natural fire to human property and

human lives. Without detailed planning, careful execution, and adequate experience, prescribed burning poses the same risks as wildfire and can have severe, negative impacts to red-cockaded woodpeckers because of fuel build-up. However, several tools now exist for reduction of these risks (e.g., Feary and Neuenschwander 1998), and if properly planned and implemented prescribed burns can be safely used to create and maintain woodpecker habitat. Benefits to the entire ecosystem far outweigh the risks associated with prescribed burning, if fires are planned and conducted with caution and guidelines are followed (see 8I).

Application of Fire to the Landscape

Aerial and ground ignition are the two most common methods used to apply fire to the landscape. Ground ignition is the more common of the two because it requires less financial resources and training. However, aerial ignition is becoming increasingly popular because more area can be burned per unit time, and the smoke dispersal is improved.

Ground ignition is accomplished by one or more techniques. Hand-held drip torches are most common, either used alone or in combination with other techniques such as mechanical torches mounted to all-terrain vehicles (ATVs). Using all-terrain vehicles increases the efficiency of ground burning operations, but entails greater safety risks than hand held torches. Caution must be exercised when using ATVs in forest stands with excessive midstory, hidden stumps, or large amounts of downed timber, and operators should be trained in vehicle use. Recently, several safety improvements have been made to ATV-mounted torches, and managers considering their use should contact state and federal agencies to learn more about these improvements.

Aerial ignition can be a very efficient method of burning large areas in a few hours. One example of a successful prescribed burning program using aerial ignition is that of the Carolina Sandhills National Wildlife Refuge (Ingram and Robinson 1998). Aerial ignition is generally accomplished through the use of a helicopter equipped with a helitorch or a plastic sphere dispenser (PSD). The helitorch uses a gel-like substance (alumi-gel) which is ignited and dispensed from a torch suspended from the helicopter. The PSD uses an apparatus mounted inside the helicopter that disperses individual spheres about 3.8 cm (1.5 in) in diameter; these spheres ignite in a few seconds once on the ground. The use of the PSD method requires a second person, other than the pilot, to operate the PSD machine. Over a thousand hectares (several thousand acres) can be burned per hour using either technique. Each technique has advantages and disadvantages; local experts should be contacted to discuss their use in various regions of the woodpecker's range.

Aerial ignition requires considerably greater protection of cavity trees than does ground ignition, because aurally ignited fires vary much more in fire intensity. If raking or mowing is used as a method of securing red-cockaded woodpecker cavity trees within an aerial-ignition burn unit, this should be done for a distance of 6.1 m (20 ft) or more from the cavity trees. Even greater distances may be required if the area has not already

undergone frequent burning and the habitat requires restoration. In this case, all clusters should be burned using ground ignition before aerial ignition of the larger burning unit.

Restoration Burning and the Reintroduction of Fire

Restoring seriously degraded habitat is perhaps the most challenging application of prescribed fire in the management of red-cockaded woodpeckers, but it can be highly successful if performed with commitment and cooperation. Wade *et al.* (1998) describe four cases in which fire has been successfully reintroduced under seemingly insurmountable circumstances: (1) reintroduction of fire to an area that was not burned for over 50 years; (2) intentional use of a high-intensity stand replacement fire; (3) burning following a major hurricane, and (4) burning within a residential subdivision. Similarly, fuel reduction and restoration of plant communities has been accomplished in many state parks in Florida (Stevenson 1998).

Restoration burns are commonly used to reduce or remove dense hardwood midstories. These burns are usually more intense than other controlled burns, and fire suppression equipment should be on site in case the fire crosses control lines. Clusters on deep, sandy soils, with a dense hardwood midstory and a sparse accumulation of ground fuels, can be effectively treated with a restoration burn during the growing season. Key to success of this management action is a thorough understanding of fire behavior in those fuel types under a variety of weather conditions. The use of fire for restoration purposes often requires burning under very specific weather parameters including those conditions identified as extreme fire weather conditions. Typically, these parameters include modest to high wind speeds, a low relative humidity, and low fuel moistures. Use of prescribed burns under these conditions requires extensive experience in the application of growing season fire and should only be attempted by experienced burners.

G. HABITAT RESTORATION

Ecological restoration is the process of returning ecosystem properties such as composition, structure, function, and dynamics to altered ecosystems. These properties are restored to within their estimated unaltered natural range of variation or, alternatively, to within ranges of variation that are capable of sustaining desired ecosystem components and processes. Thus, ecosystem restoration is rooted in the understanding and representation of natural variation in communities, ecosystems, and landscapes (White and Walker 1997). Identification of ecosystem composition, structure, function, and dynamics to be restored is achieved through the selection of appropriate reference criteria (White and Walker 1997). A variety of reference information can be derived from existing reference sites, historical data, and on-site evidence (Meffe and Carroll 1997, White and Walker 1997). However, spatial scale is important in considering natural variation. Restoration should be performed with both regional and local variation under consideration.

For red-cockaded woodpeckers, restoration of good quality habitat is vital to the recovery of the species. Loss of habitat was primary among the original causes of decline (see 1A), and the widespread increases necessary for recovery cannot be achieved without large-scale restoration of habitat. Habitat loss was caused by removal of the original old-growth forest, fire suppression, reproductive failure of longleaf pine, and conversion of longleaf and other native pine species to plantations of off-site species. Methods of site preparation have also substantially altered native groundcovers in woodpecker habitats.

Reintroduction of a fire regime patterned after historic fires is central to the restoration of native southeastern pine ecosystems—that is, habitat for red-cockaded woodpeckers. Prescribed fire should mimic the frequency, intensity, and seasonality of natural historic fire in order to maximize benefits to the fire-adapted species of southeastern pine communities. Restoration of fire to the landscape aids in restoring appropriate habitat structure and species composition. Prescribed fire facilitates the reproduction, growth, and maintenance of longleaf, shortleaf, and other native pine species, and can reestablish highly diverse native groundcovers. The restoration of these species, in turn, facilitates frequent fire—an important function—in the system. Other important management tools in habitat restoration include thinning to restore historic pine densities; protecting, planting and seeding native pines and groundcovers; and the use of site preparation methods that minimize soil disturbance.

One problem in specifying desired components and structure for ecosystem restoration is lack of information concerning historic communities and alteration of existing reference sites (White and Walker 1997, Walker 1998). Longleaf pine woodlands have been reduced to less than 5 percent of their original area, and longleaf ecosystems with intact groundcovers are even more rare (Frost 1993). Species lists and structural analyses of remnant longleaf pine ecosystems (e.g. Peet and Allard 1993, Noel *et al.* 1998) are critical. Other ecosystem types supporting red-cockaded woodpeckers, such as shortleaf and native slash pine communities, require further research attention as well. Despite these difficulties, researchers have assembled a body of information that can be used to identify general desired future conditions for southern pine ecosystems supporting red-cockaded woodpeckers. Key components of these conditions include: (1) native canopy pine species, (2) old-growth pines, (3) lower density of canopy pines than in most second and third-growth forests, and (4) healthy forb and bunchgrass groundcovers.

Restoration of Native Canopy Pines

Loss of native pines, especially longleaf but also shortleaf pine, has occurred throughout the range of red-cockaded woodpeckers. Their restoration is an important component of red-cockaded woodpecker management and recovery, primarily because these pines provide superior habitat and facilitate critical, frequent fire (Platt *et al.* 1988b). Restoration of native pine communities is a crucial aspect of ecosystem management also (see 3H). Restoration of longleaf pine has been identified as a high priority in the management of national forests. Over 40,000 ha (100,000 ac) of national

forests were restored to longleaf pine between 1988 and 1997, a 20 percent increase over 1988 levels (McMahon *et al.* 1998). An additional 140,000 ha (350,000 ac) are to be restored over the next 90 years, representing a future increase of 60 percent over 1988 levels (McMahon *et al.* 1998). Expanded use of growing-season fire is an important part of this restoration program (McMahon *et al.* 1998).

Size of Restoration Areas

An important consideration in the restoration of native pine species is the size of the area to be restored. Restoration work should not result in impacts to red-cockaded woodpeckers, either through direct loss of habitat or habitat fragmentation (Ferral 1998, F. James, pers. comm.). Clearcuts performed for this purpose should be no larger than 10 ha (25 ac), and use of smaller patches is recommended.

Restoration Methods

General information about longleaf restoration is presented in Hermann (1993) and Kush (1998), and further details can be obtained from the Longleaf Alliance (Rt. 7, Box 171, Andalusia, AL., 36420; Gjerstad *et al.* 1998). In addition, the USDA Forest Service offers information and incentives to state managers and private landowners considering the restoration of native pine species through the State and Private Forestry Programs (McMahon *et al.* 1998).

The first step in the restoration of native pines to an area currently supporting off-site species is the removal of the foreign pines (typically loblolly and slash, but also Virginia and sand pines) through small clearcuts or group selection. Site preparation methods (preferably prescribed burning) rids the area of non-merchantable pines and undesirable hardwoods while establishing proper germination conditions for desired seeds (see below and 8H for further discussions of site preparation). If no desired seedlings become established and no seed source is available, desired species must be planted or seeded. If seedlings or seeds need to be brought into the site, use of an appropriate source for the area in question is important to maintain genetic integrity and to enhance the likelihood of success (Schmidtling *et al.* 1998).

Restoration of Historic Pine Densities

Many of today's forests are densely stocked (Boyer and Farrar 1981, Landers *et al.* 1990, Noel *et al.* 1998). Density of pines in historic forests was substantially lower, as estimated from old survey data, travelers accounts, and current old-growth remnants (Foti and Glenn 1991, Masters *et al.* 1995, Noel *et al.* 1998). For example, precolonial densities for shortleaf pine forests in the Ouachita Mountains have been estimated at roughly 8.0 sq. m per ha (35 sq. ft per acre) pine basal area and 7.6 sq. m per ha (25 sq. ft per acre) of hardwood basal area (Foti and Glenn 1991, Masters *et al.* 1995). Not only are second-growth stands more dense than old-growth forests, but they typically have lower variability in density across the stand (Noel *et al.* 1998).

In the absence of active management, second-growth forests may not shift toward an old-growth structure for decades or even centuries (Noel *et al.* 1998). Second-growth longleaf forests studied by Noel *et al.* (1998) contained an overrepresentation of pines 20.3 to 40.6 cm (8 to 16 in) in dbh, and trees of these sizes were characterized by extremely low mortality and very slow growth. Thus, change in habitat structure was unlikely to occur naturally in the near future. However, researchers and managers are not always sure of the best method or methods to restore appropriate pine densities. Selective removal of small groups of trees is recommended for xeric longleaf forests, but flatwoods longleaf may require more research to develop restoration methods (Noel *et al.* 1998). Prescribed burning, patterned after the historic fire regime, can contribute to long-term restoration of appropriate pine (and hardwood) densities (Noel *et al.* 1998).

Restoration of Native Groundcovers

Longleaf pine ecosystems are famous for their highly diverse groundcovers (Walker and Peet 1983, Simberloff 1993, Peet and Allard 1993, Glitzenstein *et al.* 1998b, Walker 1998). Restoring and maintaining this diversity is a primary goal of ecological restoration in the southeast. Native groundcovers have important benefits to red-cockaded woodpeckers: native grasses are pyrogenic (Platt *et al.* 1988b, Noss 1989), and native groundcovers provide more prey than encroaching hardwoods (Provencher *et al.* 1997, 1998, Collins 1998). Also, an ecosystem approach to managing red-cockaded woodpeckers and their habitat emphasizes the conservation of native diversity.

Vegetation native to longleaf and shortleaf pine ecosystems may be best restored and maintained through the use of frequent growing season fire. Loss of groundcover diversity in the absence of fire is well documented (Christensen 1981, Ware *et al.* 1993, Peet and Allard 1993, Glitzenstein *et al.* 1998b, Walker 1998), and single fires are not sufficient to restore species diversity (Glitzenstein *et al.* 1998b). Prescribed fire is necessary to remove decades of litter accumulation and expose the mineral soil for seedling germination and early seedling growth (Walker 1998). In addition, prescribed fire opens the forest floor to sunlight, by killing off-site hardwoods and shrubs and reducing the density and stature of on-site hardwoods and shrubs. Growing season fire stimulates flowering and fruit production of native groundcover plants (Platt *et al.* 1988a, Streng *et al.* 1993). Finally, benefits of fire may be increased by restoring natural variability in the fire regime (Walker 1998).

Hardwood Control

Key to restoration of native groundcovers is the initial control of existing hardwoods. Areas long under fire suppression may require mechanical or chemical methods of hardwood removal (Conner 1989, Conner *et al.* 1995). However, such methods should be used carefully, to minimize disturbance to soils and desired native species.

Both herbicides and mechanical methods can result in increased abundance of disturbance-tolerant, ruderal species such as broom sedge (Provencher *et al.* 1998, 1999).

In a study at Eglin Air Force Base, researchers compared three hardwood reduction treatments, including the use of the commonly used herbicide, hexazinone, in a well-replicated large-scale experiment. They found that herbicide use increased disturbance-tolerant species while causing significant declines in common important species such as gopher apple (*Licania michauxii*), dwarf huckleberry (*Gaylussacia dumosa*), little bluestem (*Schizachyrium* spp.) and various legumes (e.g., Florida milk-pea, *Galactica floridana*). Some of these effects still persisted after four years and following the application of growing season fire (Provencher *et al.* 1999, L. Provencher, unpublished). Moreover, effects of herbicides on rare plant species are not well known. In a recent review of all available studies on the impacts of herbicides on vegetation, only two, including the above study, comprehensively documented the effects of herbicides across all species, including rare species (L. Provencher, unpublished).

Handtools such as chainsaws or brushhooks may have minimal impacts on native species, but use of heavy machinery should be avoided. In one study, repeated passes with a double drum chopper to remove scrub oaks (*Quercus* spp.) killed 50 percent of the existing wiregrass (Outcalt and Lewis 1990). In this same study, single passes with a light single drum chopper had little effect on groundcovers. Roller choppers may have increased effects on mesic sites (Glitzenstein *et al.* 1993).

With sufficient expertise, prescribed fire can be used to control even serious hardwood problems. Effects of fire vary with its intensity, frequency, and season, and although restoration of the historic fire regime is the desired goal, initial control of hardwoods may require manipulation of fire frequency, intensity, and season beyond those of historic fire (Robbins and Myers 1992). For example, Masters *et al.* (1995), in their recommendations for the reintroduction of fire into the shortleaf pine forests of the McCurtain County Wilderness Area in Oklahoma, called for initial use of dormant season burns to acclimate the old-growth pines to fire. These were to be followed by high frequency growing season fires to remove small stems, and then by large scale fires initiated after longer burn intervals to hasten return to precolonial conditions. To use fire successfully, managers must have solid understanding of the frequency, intensity, and season of fire necessary to achieve management objectives, and specifically identify these in the planning of a prescribed burning program.

Site Preparation

As mentioned above, mechanical and/or chemical methods of site preparation can have detrimental effects on native groundcover plants (discussed in Glitzenstein *et al.* 1993, Provencher *et al.* 1999). Effects of site preparation methods can vary depending on characteristics of the specific site, especially soil moisture content. In general, mechanical site preparation increases weedy species, and repeated use can reduce or eliminate native species. Site preparation that leads to soil disturbance will favor more ruderal, weedy, disturbance-tolerant species at the cost of sensitive species (Provencher *et al.* 1998, 1999), and recovery rates for native groundcovers may approach 50 years in xeric soils (Provencher *et al.* 1997, 1998). Windrows and other methods that create piles are among the most destructive of mechanical site preparation methods. Roller chopping

may have minimal impacts on xeric sites, especially if light machines are used (described above, Outcalt and Lewis 1990), but may be more damaging on wetter sites. Brackemounding may have the least impacts of mechanical methods, but site preparation is best performed using prescribed fire in order to minimize disturbance.

Direct Seeding and Planting

Not all of the desired plant species may return through the use of prescribed fire alone, depending on the degree of habitat alteration and the availability of natural seed sources. Progress has been made in the restoration of specific species using direct seeding and planting. For example, Hattenbach *et al.* (1998) reported successful use of direct seeding of wiregrass and several other groundcover species in the restoration of the Apalachian Bluffs and Ravines Preserve in Florida. Other examples of successful restoration of desired groundcover plants are described by Glitzenstein *et al.* (1998a, 1998b) and Bissett (1998). Researchers stress the need for frequent fire prior to and during restoration efforts to create required conditions for germination and to promote flowering. Direct seeding and planting is a labor-intensive technique conducted at very small scales. Thus, protection of existing native groundcovers should always be the first option.

H. ECOSYSTEM MANAGEMENT

Ecosystem management has been defined in many ways (reviewed by Meffe and Carroll 1997), but its various definitions contain common themes. In general, ecosystem management is an expansion of single-species or traditional management methods to include broader ecological, socioeconomic, and institutional perspectives. Meffe and Carroll (1997), in their review of ecosystem management, have developed the following composite definition:

Ecosystem management is an approach to maintaining or restoring the composition, structure, and function of natural and modified ecosystems for the goal of long-term sustainability. It is based on a collaboratively developed vision of desired future conditions that integrates ecological, socioeconomic, and institutional perspectives, applied within a geographic framework defined primarily by natural ecological boundaries.

This definition summarizes important aspects of ecosystem management common to various definitions (e.g., Grumbine 1994, Christensen *et al.* 1996), including:

1. *Conservation of biological diversity and ecological integrity.* Targets of conservation include all natural levels of organization, from genes through landscapes; the complex interactions among these levels; natural disturbance regimes; and ecosystem functions. Both natural and modified landscapes have these conservation targets.

2. *Long-term sustainability.* Sustainability, over generations and centuries, is of overwhelming importance. It should always be a clearly identified objective that is incorporated into management planning.
3. *Collaboration.* Successful ecosystem management requires cooperation among federal, state, and local agencies, tribal governments, corporations, and individuals.
4. *Desired future conditions.* Desired future conditions are determined based on historical, ecological, and cultural considerations. This vision should be specifically identified and incorporated into management planning.
5. *Ecological perspective.* Excellent science is a foundation of ecosystem management.
6. *Socioeconomic perspective.* Ecosystem management recognizes that humans are a fundamental component of the natural world, and that conservation must protect human rights as well as biological diversity. Local and indigenous people should be involved in decision-making at the outset and throughout the management process, and impacts of management actions on people must always be evaluated. Excellent social science, therefore, is also a foundation of ecosystem management.
7. *Institutional perspective.* Institutions must be flexible, to respond to changing needs and new information. Flexible administration and legislation that properly reflects human values is the third foundation of ecosystem management.
8. *Natural ecological boundaries.* Precise definitions of ecosystems are not required for ecosystem management; rather, boundaries should reflect some natural border of interest (such as a watershed or mountain range, Meffe and Carroll 1997). Therefore, ecosystem management is generally conducted at larger geographic scales than traditional management. Also, management across political boundaries can be conducted only through cooperative efforts.
9. *Adaptive management.* An important component of ecosystem management not specifically identified in Meffe and Carroll's (1997) definition is its ability to adapt to changing environmental conditions and new information. Adaptive management requires feedback from consistent and intensive biological monitoring, and indicator species must be carefully chosen to reflect management goals.

Ecosystem Management and Red-cockaded Woodpeckers

Current management for red-cockaded woodpeckers is, in many ways, an ecosystem approach. Long-term sustainability is the primary objective of management recommended in this recovery plan, and desired future conditions that will support long-term sustainability are identified herein. Cooperation among federal agencies (specifically, the U.S. Fish and Wildlife Service, the Forest Service of the U.S. Department of Agriculture, the U.S. Departments of Defense and Energy, and the National Park Service) is required in the management of core recovery populations.

Cooperation of federal, state, and local agencies, corporations, and individuals is being fostered for the management of red-cockaded woodpeckers on state and private lands. Finally, ecological borders are used for recovery units and form the basis of the translocation strategy.

Moreover, management for red-cockaded woodpeckers provides strong benefits for entire ecosystems. Such benefits are mainly the result of prescribed burning and the retention of old trees in the landscape. In addition, cavities created by red-cockaded woodpeckers or supplied to them through management are used by a host of secondary cavity species. Ecologically, single-species management of red-cockaded woodpeckers merges with ecosystem management for three main reasons: (1) red-cockaded woodpeckers are a true indicator species whose population trends mark the health of southern pine ecosystems; (2) red-cockaded woodpeckers are an umbrella species, whose protection provides simultaneous protection for many associated species; and (3) red-cockaded woodpeckers are a keystone species whose presence controls the presence and/or abundance of other species (secondary cavity users) in the community.

However, some aspects of current woodpecker management have not yet been expanded to the level of the ecosystem. One example of current management that is not consistent with an ecosystem approach is management of predation and cavity kleptoparasitism. Managers of several red-cockaded populations have instituted predator and kleptoparasite control programs, but no research has assessed the impacts of these programs on these other species. Ecosystem management protects viable populations of all native species in the region. More information concerning the population dynamics of predators and cavity kleptoparasites, and their impacts on red-cockaded woodpeckers in general, is required before methods of control can be considered part of an adaptive, ecosystem-based strategy. At present, the U.S. Fish and Wildlife Service is recommending that methods of control be non-lethal, and used only in critically small populations of red-cockaded woodpeckers (see 8E).

The primary example of current management that is not consistent with an ecosystem approach is the continued focus of most management actions, especially prescribed burning and retention of old trees, within the cluster rather than throughout the landscape. Burning and retaining old trees only in small patches provides only limited benefits to other members of southern pine communities. Moreover, such patch-based management has had detrimental effects on red-cockaded woodpeckers as well, including decreased value of foraging habitat (James *et al.* 1997, Walters *et al.* 2000), increased cavity damage by pileated woodpeckers (Saenz *et al.* 1998), and increased mortality of cavity trees due to pests such as southern pine beetles (Conner *et al.* 1997a). Fundamental change in the scale of prescribed burning and beneficial silvicultural practices is required for both ecosystem management and the recovery of red-cockaded woodpeckers.

However, some management actions must continue to be applied at the level of individual territories or aggregations of territories rather than at a landscape scale. That is, some aspects of single-species management continue to be critical to the recovery of

red-cockaded woodpeckers. Chief among these are cavity management (see 8C), establishment of strategically placed recruitment clusters (8A), and translocation (8F). Predator and cavity kleptoparasite control is a single-species management technique also, but it differs from those listed above in that it can potentially disrupt natural ecosystem processes and impact other native species.

Thus, at present red-cockaded woodpeckers are best managed with a combination of single-species and ecosystem management techniques, and other members of southern pine communities benefit substantially from such management. Once red-cockaded woodpeckers attain recovery, single-species methods will not be required; ecosystem management by itself, including continued monitoring of red-cockaded woodpeckers, will provide long-term sustainability for all members of southern pine communities.

4. CURRENT STATUS AND CONSERVATION INITIATIVES

A. PRIVATE LANDS

Conservation of red-cockaded woodpeckers on privately owned lands is an important part of species recovery (Costa 1995b, 1997, Bonnie and Bean 1996, Bonnie 1997), although primary support for recovery is provided by federal properties (4C). Groups and populations of red-cockaded woodpeckers on private lands have substantial value as reservoirs of genetic resources, sources of immigration for other populations, and as stepping stones to facilitate dispersal between other populations. In addition, woodpeckers on private lands have inherent ecological, cultural, and historical value. However, other than the restriction against removing existing red-cockaded woodpeckers, their foraging habitat, and cavity trees, there is no federal law that requires private landowners to participate in active conservation. Thus, the role of private landowners in species recovery is important but voluntary.

The voluntary nature of active conservation on the part of private landowners has some benefits. Private lands conservation arising from local participants can be more meaningful and longer lasting than attempts at regulating private land use by federal authorities. The most successful conservation programs are those that strike a balance between voluntary participation and federal control. For endangered species, private landowners require a mechanism for resolving land use conflicts, but mitigation must be adequate and federally supervised (Bean and Wilcove 1997). Flexibility, with appropriate boundaries, can foster genuine conservation interest on the part of local landowners and reduce the resentment that is a common result of enforcement of federal regulations (Bean and Wilcove 1997, Bonnie 1997). For example, volunteer participants in Safe Harbor programs (below) have shown increased concern for red-cockaded woodpeckers on their lands (Bonnie 1997). Education, incentives, and the removal of disincentives are key factors facilitating the rise of conservation among private individuals (USFWS 1979, Bonnie and Bean 1996, Kennedy *et al.* 1996).

These benefits of voluntary conservation were recognized, encouraged, and incorporated into a private lands conservation strategy by the U.S. Fish and Wildlife Service during the 1990's (Costa 1995b; described below). Some early efforts may have fallen short of conservation goals (Bonnie 1997), but with continual improvements the private lands conservation strategy of the U.S. Fish and Wildlife Service has shown remarkable success.

The Endangered Species Act and Private Landowners

Federal law does not require private landowners to participate in the recovery of threatened and endangered species but does prohibit their 'take' (Section 9a of the Act). The term, take, means to "harass, harm, pursue, hunt, shoot, wound, kill, trap, capture, or collect, or to attempt to engage in any such conduct" (Section 3.18). Habitat destruction and alteration are considered forms of take, following a Supreme Court ruling on this issue (Sweet Home vs. Babbitt). The Endangered Species Act does provide a mechanism for take of endangered species on private lands if that take is "incidental to, and not the purpose of, the carrying out of an otherwise lawful activity" (Section 10a of the Act). Incidental take may be permitted by the U.S. Fish and Wildlife Service only after the applicant submits a detailed Habitat Conservation Plan (HCP) that includes steps to be taken to minimize and mitigate impacts from the proposed actions (Section 10a). Thus, the U.S. Fish and Wildlife Service has formulated guidelines for mitigation of impacts to red-cockaded woodpecker groups (below). Still, incidental take permits are issued rarely, because generally alternatives to incidental take exist, and the Act requires the evaluation of alternatives and their use if appropriate (Section 10a). Federal properties are not involved in the incidental take permitting process, but rather must consult with the U.S. Fish and Wildlife Service on proposed actions that may have the potential to result in incidental take (Section 7a).

Recent Trends and Current Status

Despite continued protection under the Endangered Species Act, the decline and local extirpation of red-cockaded woodpeckers on private lands has been well documented across their range. Reports from North Carolina (Carter 1974, 1990, Carter *et al.* 1983, 1995), South Carolina (Cely and Ferral 1995), Georgia (Baker 1981, 1995), Arkansas (James and Neal 1989), Texas (Ortego and Lay 1988), Florida (Baker 1983), and range wide (Thompson 1976, Ligon *et al.* 1986, James 1995) show declines and local extirpations into the early 1990's. These losses are the result of a variety of factors including loss and fragmentation of habitat, fire suppression and resultant changes in habitat structure, and vulnerability to environmental and demographic stochasticity because of small population size. Currently, there are roughly 1000 known active clusters on private lands in 11 states (Costa and Walker 1995), and the existence of up to 250 additional groups is considered likely.

The Private Lands Conservation Strategy

The private lands conservation strategy was developed by the U.S. Fish and Wildlife Service in response to the realization that red-cockaded woodpeckers on private lands were important to the recovery of the species, and that their loss was a significant biological problem (Costa 1995b, 1997). Moreover, the U.S. Fish and Wildlife Service recognized that conservation of red-cockaded woodpeckers on private lands would require a multi-faceted approach based on conservation science and innovative conservation partnerships (Costa 1995b, 1997). The strategy has been aggressively implemented, modified as necessary based on new scientific findings, and regularly evaluated to insure goals are being achieved. Five primary objectives of the private lands strategy are to (1) increase the acreage of private land under management for red-cockaded woodpeckers; (2) maintain or increase the larger populations on private lands, (3) establish healthy, spatially aggregated, and protected groups of woodpeckers to offset losses, (4) foster and develop corporate partnerships between and among federal, state, and private parties responsible for and interested in red-cockaded woodpecker recovery and (5) increase, via translocation, the size of populations on state and federal lands (Costa 1995b). This last objective does not imply that federal properties are mitigation sites, but private lands do contribute birds to public properties as part of the regional translocation strategy.

The implementation of the private lands strategy between the U.S. Fish and Wildlife Service and private land conservation partners since 1992 has helped to slow, stabilize, and in some cases reverse population declines among woodpeckers on privately owned lands. It has resulted in significantly increased protection for many woodpecker groups and their habitat on privately owned lands, and raised the possibility that such protection can become the normal standard rather than the exception. Finally, the private lands strategy has resulted in the creation of strong and effective partnerships with a multitude of diverse partners. Currently, 355 red-cockaded woodpecker groups on 83,352 ha (205,962 ac) of private lands are protected. Protection for 92 groups, on 3966 ha (9,800 ac), is pending under draft agreements with the U.S. Fish and Wildlife Service. Thus, current and pending agreements provide protection for 44 percent of the known red-cockaded woodpeckers on private lands. Additionally, several landowners in signed and pending agreements have agreed to increase their existing populations. These increases could result in 71 additional groups, a 20 percent increase over current numbers protected.

The development of the private lands strategy began in the early 1990's, with initial attempts to protect woodpeckers on forest industry lands (Costa 1995b, Wood and Kleinhofs 1995). In 1992, the first Memorandum of Agreement (below) was signed with Georgia Pacific in an attempt to protect approximately 90 groups in Arkansas/Louisiana. Seven other Memoranda of Agreement followed (Costa 1997). These are 'no-take' agreements under which a corporation agrees to protect occupied habitat and conduct some habitat management (Bonnie 1997, Costa 1997). Since 1995, the U.S. Fish and Wildlife Service has shifted from Memoranda of Agreement to Habitat Conservation Plans (HCP's; Bonnie 1997, Costa 1997), in which incidental take of existing and/or

future woodpecker groups is permitted in exchange for management of occupied and unoccupied habitat. Habitat Conservation Plans, authorized under Section 10 of the Endangered Species Act, can involve a variety of landowners, including timber and other corporations, private citizens, and developers. Three forms of HCP's currently exist: individual plans, statewide plans, and Safe Harbor Agreements.

Memoranda of Agreements

Memoranda of Agreement are legal conservation agreements between the U.S. Fish and Wildlife Service and corporate landowners. The agreement outlines management actions by which the corporation can satisfy responsibilities under the Endangered Species Act and the U.S. Fish and Wildlife Service's guidelines for habitat management, and meet corporate objectives for land management. These management actions typically include population monitoring, management and retention of current and future nesting habitat, maintenance of adequate foraging habitat, and research and educational initiatives. Several MOA's also include state or other federal agencies as cooperators. Motivation to enter into such agreements includes reduced risk of litigation, prestige and satisfaction associated with conservation efforts, and consolidation of populations and responsibility (Costa and Edwards 1997). Currently, over 12,990 ha (32,100 ac) of habitat and 83 active woodpecker clusters are managed under Memoranda of Agreement.

Individual Habitat Conservation Plans

Individual Habitat Conservation Plans allow the 'incidental take' of red-cockaded woodpecker groups with mitigation, as authorized under the Endangered Species Act. Both the plan and the associated mitigation are funded by the landowner. Early HCP's for individual landowners were criticized because the mitigation required was not considered sufficient to offset the permitted loss of groups (Bonnie 1997). These critics correctly identified two major faults of early mitigation efforts. First, occupation of the newly established clusters was not assured. Second, the creation of clusters on federal properties did not truly mitigate damage to privately owned clusters, because federal agencies are already required to conserve (recover) their populations. In response to these criticisms, the current policies governing the use of mitigation (below) require that one occupied cluster be established for each active cluster harmed or removed. In addition, new groups are established on private lands when possible (below).

Since 1995, the U.S. Fish and Wildlife Service has authorized ten incidental take permits for non-industrial forest landowners. Under these permits, 27 groups of red-cockaded woodpeckers may be impacted or removed, pending completion of mitigation. Mitigation for these groups includes the probable establishment of 52 new groups through creation of recruitment clusters and/or translocation of juveniles to unoccupied clusters (Costa 1997).

The U.S. Fish and Wildlife Service has also issued two individual HCP's for industrial forest landowners. These plans provide for the long-term protection of 52 groups of red-cockaded woodpeckers.

Statewide Habitat Conservation Plans

A statewide Habitat Conservation Plan (not including statewide Safe Harbor, below) permits the incidental take of demographically isolated groups only. At the present time, biologists and managers are working to develop a reasonable definition of demographic isolation for the purpose of statewide HCP's. It is known that isolation of red-cockaded woodpecker groups results in decreased likelihood of group survival. However, research into the isolation of groups has been designed to identify spatial arrangements that increase population persistence, not to identify a statewide standard for incidental take (Bonnie 1997). Establishing a threshold measure of isolation above which groups would be available for statewide incidental take is a matter of some debate, and requires further research attention.

Safe Harbor

The Safe Harbor program has been an immense success for both landowners and red-cockaded woodpeckers (Bonnie 1997, Costa 1997, 1999). First established in 1995 in the Sandhills region of North Carolina (Costa and Kennedy 1996), Safe Harbor programs have now been implemented statewide in South Carolina, Texas, and Georgia. Louisiana, Alabama, Florida, North Carolina and Virginia are working on similar plans. Under a Safe Harbor agreement, a landowner agrees to actively manage nesting and foraging habitat (i.e., a safe harbor) for the number of active red-cockaded woodpecker clusters equal to those present when the agreement is initiated. Landowners must also agree to enhance existing habitat and/or improve additional potential woodpecker habitat, typically through the use of prescribed fire and cavity management. In turn, the landowner receives an incidental take permit, authorizing a land use change, for any additional woodpecker groups that may occupy the property in the future as a result of beneficial management practices. Thus, private landowners are free to manage their properties with prescribed fire, thinnings, lengthened timber rotations, or other actions that may benefit red-cockaded woodpeckers without fear of additional land-use restrictions.

Landowners enrolled in Safe Harbor agreements may choose to enter into mitigation banking (below), and increase their resident populations in safe harbor in exchange for a mitigation fee. This can be a powerful incentive for private landowners to join a Safe Harbor program and aggressively manage their lands for red-cockaded woodpeckers (Bonnie and Bean 1996, Costa and Kennedy 1996, Kennedy *et al.* 1996). Mitigation banks can be established only by following the guidelines presented below.

As of 2000, 157 groups, 34 landowners, and 48,372 ha (119,526 ac) in South Carolina, 50 groups, 40 landowners, and 11,969 ha (29,576 ac) in North Carolina and 14 groups, 2 landowners, and over 2,800 ha (7000 ac) in Texas enrolled in Safe Harbor

agreements (USFWS, unpublished). Many of these groups provide important support for nearby recovery populations.

Mitigation

No Net Loss of Groups

The philosophy guiding mitigation policy is that there be no net loss of red-cockaded woodpecker groups, and a primary objective is to assure that the status of the species as a whole is better following mitigation than before. Mitigation of impacts to red-cockaded woodpeckers is generally achieved through the establishment of a woodpecker group in another location, for every group that is affected by the proposed action. The minimum required ratio of newly established to impacted groups is one to one. For the ten HCP permits issued to date, this ratio has been two to one (Costa 1997). Preservation credits, discussed below, are an exception to the required one to one ratio.

Mitigation Site

The location in which new groups are established is known as the mitigation site. This term refers to both the actual recruitment clusters and the population that contains the newly established groups. Four factors are important to the choice of mitigation sites: geographic location, ownership class (i.e., prior commitment to recovery), degree of protection in place, and amount of available habitat (i.e., maximum future population size). Local mitigation is preferred and mitigation within the recovery unit is required; that is, the mitigation site should be within the same local area but must be within the same recovery unit as the impacted groups. These geographic restrictions serve ecological goals and reduce costs.

The first priority for ownership class of mitigation sites is private and state lands. When all opportunities to mitigate on private and state lands within the above geographic restrictions have been exhausted, federal lands shall be considered. Mitigation on federal properties will be conducted only if it is the sole appropriate option within the recovery unit. In general, the use of federal properties as mitigation sites for impacts on private lands is strongly discouraged. Additionally, the U.S. Fish and Wildlife Service prefers that mitigation sites have a degree of protection similar to or greater than that of impacted groups.

Mitigation sites must have sufficient habitat to support at least 10 groups of red-cockaded woodpeckers in territories that are aggregated, not isolated, in space. Only with a highly aggregated spatial structure do populations of 10 woodpecker groups have any reasonable chance of persisting over periods of 20 years or more (Letcher *et al.* 1998). Therefore, the mitigation site must be able to support an average group density of 4 groups per 259 ha (4 groups per sq. mi), and must consist of continuous pine habitat without dispersal barriers. Only habitat meeting the definition of foraging habitat for respective ownership categories (see 8G) may be counted as continuous pine habitat.

Mitigation Groups

Mitigation groups are those newly established in exchange for permission to impact groups, on a one-to-one basis as discussed above. Mitigation groups must have equivalent breeding status as impacted groups. In other words, if an impacted group consists of a solitary male, then only a solitary male needs to be established for mitigation, but if an impacted group consists of a potential breeding group, then a potential breeding group must be established as the mitigation group. Helpers do not need to be “replaced”.

Mitigation groups are typically established prior to the impact on existing groups. However, incidental take may occur prior to successful mitigation if legally binding implementation agreements and performance bonds are in place. A mitigation group is considered established if evidence of breeding is detected or if the same potential breeding group or solitary male remain in the mitigation cluster for six months including a breeding season (normally, the breeding season includes April, May, and June).

Tools for Mitigation

Several tools to facilitate mitigation exist, including mitigation credits, mitigation banks, and preservation credits. A mitigation credit is earned once a mitigation group has been established (one credit is equal to one group), and is used by impacting an existing group. A mitigation credit can be used immediately after earning or stored in a mitigation bank to be used in the future. Mitigation credits stored in a bank can also be made available for sale to third parties requesting a permit to impact an existing group or groups. A mitigation bank is the mitigation site in which new groups are established. Guidelines for mitigation sites (above) apply to mitigation banks. Mitigation banks may be owned by a single or multiple landowners, but must have approved habitat management plans including regular prescribed burning and cavity management in place.

Finally, a preservation credit is earned by increasing the protection of one to three existing groups in exchange for the incidental take of one group. Increased protection may take the form of private land conservation easements, direct land acquisition, and subsequent transfer to protected/managed public land agencies or other conservation programs that ensure protection, but must be in place for perpetuity. In addition, preservation groups must benefit from population monitoring (Level IV, see 3A) and habitat management, including frequent prescribed burning (8I), cavity and cluster management (8C, 8D), and provision of foraging habitat that meets the recovery standard (8G). Perpetual protection of one to three groups in excellent habitat in exchange for the loss of one group is considered an improvement in the conditions faced by red-cockaded woodpeckers as a whole, in agreement with the overall objective of mitigation policy.

The specific ratio for preservation credits is determined on a case-by-case basis. Variables used to calculate this ratio include location, population size, trend, viability, and ownership, forest type, breeding status, and available foraging habitat. The final ratio is based on a careful comparison of the status of these variables for both the

impacted population and the mitigation site. These variables are used to ensure that the biological value of the group being impacted is replaced or improved upon by the mitigated group.

Funding for Mitigation

Mitigation is funded by the landowner performing the action that will impact woodpecker groups. Mitigation costs include a management endowment sufficient to cover habitat management, such as prescribed burning, for the mitigation groups for 5 years (one full generation for red-cockaded woodpeckers). Other costs include the initial provisioning of cavities and initial midstory control in the recipient cluster as well as the costs of translocating juvenile birds to create mitigation groups and translocating resident adults from affected clusters upon successful mitigation.

Other Incentive Programs

Several programs other than Safe Harbor Agreements are available to assist private landowners in management of their lands, but unlike Safe Harbor these are not designed directly for red-cockaded woodpeckers. However, programs that could potentially benefit woodpeckers are available through the Farm Services Agency, Natural Resources Conservation Service, U.S. Fish and Wildlife Service, and state forestry agencies. Local offices of the administering agency or organization should be contacted for information about future sign-ups and eligibility requirements.

Farm Services Agency

The Conservation Reserve Program offers annual rental payments and cost-share assistance to plant permanent areas of grass and trees on land that is subject to erosion, and to improve soil, water, and wildlife resources. Assistance for up to 50 percent of costs is available for the 10 to 15 year contracts. This program is most applicable to agricultural lands. However, some management practices implemented under these programs could benefit red-cockaded woodpeckers.

Natural Resources Conservation Service

Landowners who participate in the Wetlands Reserve Program may sell a conservation easement or enter into a cost-share restoration agreement to restore and protect wetlands. Landowners receive financial incentives to enhance wetlands in exchange for retiring marginal agricultural land. In addition to farmland, eligible lands include production forestland where hydrology has been altered, riparian areas that link protected wetlands, and lands adjacent to protected wetlands that contribute significantly to wetland functions and values. The program offers landowners three options: permanent easements, 30-year easements, and restoration cost-share agreements of at minimum 10-year duration. Landowners continue to control access to the land—and may lease the land—for hunting, fishing, and other recreational activities requiring no development.

The Wildlife Habitat Incentives Program is designed to help private landowners develop and improve wildlife habitat on their lands. Participating landowners work with the Natural Resources Conservation Service to prepare a wildlife habitat development plan in consultation with the local conservation district. The plan describes the landowner's goals for improving wildlife habitat, a list of practices, a schedule for installing them, and steps necessary to maintain the habitat for the life of the agreement. The participant enters into a cost-share agreement usually lasting at least 10 years. The landowner agrees to maintain the cost-shared practices and allows monitoring to judge the effectiveness of the practice. The U.S. Department of Agriculture agrees to provide technical assistance and pay up to 75 percent of the cost of identified practices.

The Environmental Quality Incentives Program is for farmers and ranchers who face serious threats to soil, water, and related natural resources. The program offers financial, educational, and technical help to install or implement structural, vegetative, and management practices called for in 5 to 10-year contracts. Eligible lands include cropland, rangeland, pasture, forestland, and other farm or ranch lands where the program is delivered. Cost-sharing may provide up to 75 percent of the funds for certain conservation practices.

The Forestry Incentives Program is intended to assure the nation's ability to meet future demand for sawtimber, pulpwood, and quality hardwoods. The program pays cost sharing of up to 65 percent (with a limit of \$10,000 per person per year) for tree planting, timber stand improvement, and site preparation for natural regeneration. The state forester provides technical advice in developing a management plan and helps find approved vendors, if needed, for completing the work. Private, non-industrial landowners who own less than 4,047 ha (1,000 ac) are eligible to participate in the program. However, this program is available only in selected counties.

U.S. Fish and Wildlife Service

The Partners for Wildlife Program provides technical and financial assistance to private landowners that are restoring and enhancing fish and wildlife habitat. Program emphasis is on restoration of historic vegetation and hydrology. Seventy percent of the project area must reflect the historic vegetation and hydrology while 30 percent may consist of wildlife enhancement activities. Landowners must sign a minimum of 10-year agreement for some projects, and a 25-year agreement for restoration projects.

State Forestry Agencies

The Forestry Stewardship Program is intended to stimulate management of non-industrial, private forest land using multiple-use concepts. This technical assistance program provides management recommendations to fit the landowner's objectives for forest management. Wildlife habitat, water quality, and soil protection are examples of objectives that can be incorporated into the landowner's management plan. Cost-share is available through the Stewardship Incentives Program, which will reimburse landowners

for 75 percent of the cost of certain forest management practices, including those intended to improve habitat for endangered species.

State incentive programs administered by the respective state forestry agencies often emphasize reforestation. Through reforestation, however, other objectives of the landowner, such as creation or enhancement of habitat for red-cockaded woodpeckers, can be addressed. Some state wildlife agencies also administer incentive programs. Examples include Kentucky's Habitat Improvement Program and Arkansas' Acres for Wildlife Program. Not all state forestry or wildlife agencies within the range of the red-cockaded woodpecker offer incentive programs.

B. STATE LANDS

Status and Distribution

As of 1998, there were an estimated 520 active clusters of red-cockaded woodpeckers in 42 populations on state lands in eight states (USFWS, unpublished, see Tables 2 and 3). Largest concentrations of woodpeckers on state lands occur in Florida, North Carolina, and South Carolina. Six states contain woodpeckers on federal and/or private lands, and none on state lands (Table 2).

During the 1970's, Jackson (1978b) found that approximately 300 clusters, or 8.6 percent of all reported clusters, were located on lands owned by state or local governments. These clusters were distributed across ten states, with the largest concentrations occurring in Florida and South Carolina. Seven of the remaining eight states had less than 12 clusters on state or local lands. Although it is clear that several states had, by 1998, lost all woodpeckers on state lands, comparison of current population sizes with those from the 1970's is hampered by inconsistent survey techniques and increasing survey effort across time. (Cely and Ferral 1995, Ortego *et al.* 1995, J. Cely, pers. comm.).

Conservation of woodpeckers on state lands is improving, but much progress remains to be made. Habitat management plans, including population goals, have not yet been established for all state lands (see Table 3). Through interviews with state land managers and biologists, J. Hovis (unpublished) found that most state agencies have implemented a prescribed burning regime on their lands inhabited by red cockaded woodpeckers. Beyond this, however, the level of management and population monitoring varies considerably both within and among states. For example, some state lands have never been surveyed completely for cavity trees, whereas others have been surveyed but the demography of the resident red-cockaded woodpecker population is unknown. Today, only a few populations on state lands have been intensively managed and/or monitored on a long-term basis. These include the McCurtain County Wilderness Area in Oklahoma (M. Howery, pers. comm.), the Sandhills Game Lands in North Carolina (Walters *et al.* 1988a), and the Sand Hills State Forest in South Carolina (Ferral 1998).

Recovery Role

State lands can contribute to the recovery of the red-cockaded woodpecker in numerous ways. Some state lands will contribute by being part of a designated recovery population. For example, the Holly Shelter Game Lands in North Carolina is part of a primary core population in the Mid-Atlantic Coastal Plain Recovery Unit, and the Sand Hills State Forest of South Carolina is part of the secondary core population in the Sandhills Recovery Unit (see 7). Other state lands will contribute as essential support populations (see 7), with a role in recovery equivalent to that of federal lands. These designated recovery populations are in central and south Florida, a unique region in the southernmost part of the species range where there are no primary and secondary core populations. The remaining state lands contribute to the conservation and recovery of red-cockaded woodpeckers as significant and important support populations (see 7).

Finally, state lands can contribute to recovery as mitigation sites (see 4A). Through the mitigation process, red-cockaded woodpecker populations on state lands could be enhanced or restored. Establishing state lands as mitigation sites, however, would require a commitment from the state agencies involved to monitor and manage their woodpecker populations on a long-term basis. Unfortunately, many state agencies have neither the personnel nor funds required to fill such a commitment. Although mitigation monies could be used to finance some management and monitoring activities, long-term programs on state lands will require additional funding. Accordingly, state agencies should be encouraged to seek Sections 6 funds through the U.S. Fish and Wildlife Service to initiate or enhance their activities on state lands with red-cockaded woodpeckers.

TABLE 2. Estimated number of active red-cockaded woodpecker clusters in 1998 on lands owned by state governments (USFWS, unpublished), compared to estimated potential number of active clusters on lands currently managed for red-cockaded woodpeckers (sum of identified population goals, or, if no goal is available, the current size). See Table 3 for list of individual properties currently supporting red-cockaded woodpeckers and more information concerning population goals.

State	Active Clusters (1998)	Potential Active Clusters (Pop. Goals)	Number of Properties (i.e., populations)
Alabama	0	0	0
Arkansas	1	1	1
Florida	186	324	13
Georgia	2	2	1
Kentucky	0	0	0
Louisiana	0	0	0
Mississippi	1	1	1
North Carolina	163	201	7
Oklahoma	11	44	1
South Carolina	137	232	14
Texas	19	19	3
Virginia	0	0	0
Total	520	824	42

TABLE 3. Status of red-cockaded woodpecker populations on public lands, by state. Estimated current size, in number of active clusters during breeding season 1999 (or 1998 in some cases), is given for all populations and subtotaled by state. Population goal, taken from management plans and based on available habitat (81 ha/group; 200 ac/group), is given where available; NA indicates management plans not yet available. Total of population goals represents the estimated maximum potential number of red-cockaded woodpeckers that current public land base managed for red-cockaded woodpeckers can support, assuming 1 group per 81 ha (200 ac; higher densities are possible). For sums of goals, current size is used if goal is unavailable.

State	Population	Recovery Unit	Owner-ship	Size (#active clusters)	Pop. Goal (#active clusters)
AL	Oakmulgee RD	Upper East Gulf Coastal Plain	Federal	123	394
AL	Talladega/Shoal Creek RD's	Cumberlands	Federal	3	413
<i>subtotal</i>				126	807
AR	Felsenthal NWR	Upper West Gulf Coastal Plain	Federal	15	47
AR	Ouachita NF	Ouachita Mountains	Federal	16	400
<i>subtotal</i>				31	447
FL	Apalachicola RD	East Gulf Coastal Plain	Federal	486	500
FL	Avon Park Air Force Range	South/Central Florida	Federal	21	50
FL	Big Cypress National Preserve	South/Central Florida	Federal	40	73
FL	Blackwater River SF	East Gulf Coastal Plain	State	20	30
FL	Cecil M. Webb WMA	South/Central Florida	State	27	NA
FL	Conecuh NF	East Gulf Coastal Plain	Federal	14	309
FL	Eglin AFB	East Gulf Coastal Plain	Federal	295	500
FL	Goethe SF	South/Central Florida	State	26	150
FL	Howe Scott Preserve	South/Central Florida	State	1	NA
FL	J. W. Corbett WMA	South/Central Florida	State	8	NA
FL	Ocala NF	South/Central Florida	Federal	18	179
FL	Osceola NF	South Atlantic Coastal Plain	Federal	63	462
FL	St. Marks NWR	East Gulf Coastal Plain	Federal	6	71
FL	St. Sebastian River State Buffer Preserve	South/Central Florida	State	9	NA
FL	Tate's Hell SF	East Gulf Coastal Plain	State	25	NA
FL	Three Lakes WMA	South/Central Florida	State	35	NA
FL	Wakulla RD	East Gulf Coastal Plain	Federal	125	500
FL	Withlacoochee SF	South/Central Florida	State	50	130
<i>subtotal</i>				1240	2601
GA	Fort Benning	Sandhills	Federal	186	450
GA	Fort Gordon	Sandhills	Federal	3	25
GA	Fort Stewart	South Atlantic Coastal Plain	Federal	198	500
GA	Laura S. Walker SP	South Atlantic Coastal Plain	State	2	NA
GA	Oconee NF	Piedmont	Federal	17	250
GA	Okefenokee NWR	South Atlantic Coastal Plain	Federal	29	127
GA	Piedmont NWR	Piedmont	Federal	37	96
<i>subtotal</i>				469	1423
KY	Daniel Boone NF	Cumberlands	Federal	7	66
<i>subtotal</i>				7	66

Table continued next page.

TABLE 3 (cont.). Status of red-cockaded woodpecker populations on public lands, by state.

State	Population	Recovery Unit	Owner-ship	Size (#active clusters)	Pop. Goal (#active clusters)
LA	Alexander SF	West Gulf Coastal Plain	State	5	5
LA	Black Bayou NWR	West Gulf Coastal Plain	Federal	1	NA
LA	Big Branch Marsh NWR	*	Federal	9	NA
LA	Calcasieu RD (Vernon)	West Gulf Coastal Plain	Federal	146	302
LA	Catahoula RD	West Gulf Coastal Plain	Federal	31	328
LA	D'Arbonne NWR	Upper West Gulf Coastal Plain	Federal	4	5
LA	Evangeline NF	West Gulf Coastal Plain	Federal	72	231
LA	Fort Polk	West Gulf Coastal Plain	Federal	44	179
LA	Kisatchie RD	West Gulf Coastal Plain	Federal	38 ¹	296
LA	Peason Ridge	West Gulf Coastal Plain	Federal	27	120
LA	Upper Ouachita NWR	Upper West Gulf Coastal Plain	Federal	1	1
LA	Winn RD	West Gulf Coastal Plain	Federal	16	263
<i>subtotal</i>				394	1740
MS	Bienville NF	Upper East Gulf Coastal Plain	Federal	106	500
MS	Biloxi RD	East Gulf Coastal Plain	Federal	6	368
MS	Chickasawhay RD	East Gulf Coastal Plain	Federal	13	502
MS	Homochitto NF	East Gulf Coastal Plain	Federal	45	254
MS	Noxubee NWR	Upper East Gulf Coastal Plain	Federal	38	88
<i>subtotal</i>				208	1712
NC	Alligator River NWR	Mid-Atlantic Coastal Plain	Federal	2	20
NC	Bladen Lakes SF	Mid-Atlantic Coastal Plain	State	3	NA
NC	Camp Mackall	Sandhills	Federal	11	12
NC	Croatan NF	Mid-Atlantic Coastal Plain	Federal	60	135
NC	Dare Co. Bombing Range	Mid-Atlantic Coastal Plain	Federal	6	46
NC	Fort Bragg	Sandhills	Federal	350	350
NC	Holly Shelter Game Lands	Mid-Atlantic Coastal Plain	State	38	38
NC	MCB Camp LeJeune	Mid-Atlantic Coastal Plain	Federal	50	184
NC	McCain Tract	Sandhills	Federal	5	NA
NC	Military Ocean Terminal Sunny Point	Mid-Atlantic Coastal Plain	Federal	6	17
NC	Pee Dee NWR	Piedmont	Federal	1	10
NC	Pocosin Lakes NWR	Mid-Atlantic Coastal Plain	Federal	4	NA
NC	Sandhills Game Lands	Sandhills	State	132	160
NC	Weymouth Woods SP	Sandhills	State	6	NA
<i>subtotal</i>				674	990
OK	McCurtain Co. Wilderness Area	Ouachita Mountains	State	12	44
<i>subtotal</i>				12	44
SC	Carolina Sandhills NWR	Sandhills	Federal	118	200
SC	Charleston Naval Weapons Station	South Atlantic Coastal Plain	Federal	2	12
SC	Cheraw State Fish Hatchery	Sandhills	State	1	1

Table continued next page.

TABLE 3 (cont.). Status of red-cockaded woodpecker populations on public lands, by state.

State	Population	Recovery Unit	Owner-ship	Size (#active clusters)	Pop. Goal (#active clusters)
SC	Cheraw SP	Sandhills	State	7	20
(cont.)					
SC	Fort Jackson	Sandhills	Federal	21	126
SC	Francis Marion NF	Mid-Atlantic Coastal Plain	Federal	334	450
SC	Hampton Plantation Park	Mid-Atlantic Coastal Plain	State	1	NA
SC	Lewis Ocean Bay Heritage Preserve	Mid-Atlantic Coastal Plain	State	3	10
SC	Longleaf Pine Heritage Preserve	Mid-Atlantic Coastal Plain	State	3	NA
SC	Manchester SF	Sandhills	State	3	NA
SC	Persanti Island	South Atlantic Coastal Plain	State	3	NA
SC	Poinsett Weapons Range	Sandhills	Federal	6	30
SC	Sand Hills State Forest	Sandhills	State	50	145
SC	Sandy Island	Mid-Atlantic Coastal Plain	State	36	NA
SC	Santee Coastal Reserve	Mid-Atlantic Coastal Plain	State	9	NA
SC	Santee SP	South Atlantic Coastal Plain	State	1	NA
SC	Savannah River Site	South Atlantic Coastal Plain	Federal	31	418
SC	Webb WMA	South Atlantic Coastal Plain	State	11	NA
SC	Wedge Plantation	Mid-Atlantic Coastal Plain	State	1	NA
SC	Yawkey Wildlife Center	Mid-Atlantic Coastal Plain	State	8	NA
<i>subtotal</i>				649	1488
TX	Angelina NF	West Gulf Coastal Plain	Federal	30	252
TX	Davy Crockett NF	West Gulf Coastal Plain	Federal	51	330
TX	L. D. Fairchild SF	Upper West Gulf Coastal Plain	State	3	7
TX	Sabine NF	West Gulf Coastal Plain	Federal	25	462
TX	Sam Houston NF	Upper West Gulf Coastal Plain	Federal	168	541
TX	W. G. Jones SF	Upper West Gulf Coastal Plain	State	14	14
<i>subtotal</i>				291	1606
TOTAL²				4133	13409

*Big Branch Marsh National Wildlife Refuge is not located in a recovery unit. It is the sole population found in the Gulf Coast Prairies and Marshes ecoregion.

¹Estimated maximum number of active clusters, May 2000.

²Differs from totals in Table 5 because Piney Grove Nature Preserve, privately owned by The Nature Conservancy, is not included in this table.

C. FEDERAL LANDS

Conservation of red-cockaded woodpeckers as a species depends primarily on the conservation of populations on federal lands, for several reasons. First, the vast majority of red-cockaded woodpeckers in existence today are on federal lands (Costa and Walker 1995, James 1995). Second, federal properties contain most of the land that can reasonably be viewed as potential habitat for red-cockaded woodpeckers (USFWS 1985). Third, existing legislation, especially the Endangered Species Act (Section 7) but also the National Forest Management Act and others, require that federal agencies conserve listed species and maintain biodiversity within their lands. In the Endangered Species Act (Section 3), conservation is defined as “the use of all methods and procedures necessary to bring an endangered species or threatened species to the point at which the measures provided pursuant to this act are no longer necessary.” Thus, to the extent that legislation reflects public perception, it is the public’s view that recovery of endangered species and conservation of biodiversity is a responsibility of the federal government to be conducted primarily on publicly owned lands under federal control. This is a difficult task, as it requires the protection of biodiversity at or near precolonial levels on minute remnants of the habitat base. Private landowners can contribute substantially to conservation, but such contributions above the required protection against direct harm (take) are voluntary (see 4A).

Federal properties supporting populations of red-cockaded woodpeckers include national forests, military installations, national wildlife refuges, a national preserve, and a Department of Energy property. National forests support the majority of core woodpecker populations required for delisting and therefore have a uniquely important role in the recovery of red-cockaded woodpeckers. Second to national forests in recovery importance are the military installations. National wildlife refuges have a smaller but important role in woodpecker recovery, as do the remaining occupied federal properties.

National Forests

Current Status and Trends

Currently, there are 25 populations of red-cockaded woodpeckers on national forests (see map insert and Table 4), ranging in size from 3 and 7 active clusters (Talladega/Shoal Creek Ranger Districts of the Talladega National Forest and Daniel Boone National Forest, respectively) to 486 active clusters (Apalachicola Ranger District, Apalachicola National Forest). The population on the Apalachicola Ranger District is one of the two largest remaining woodpecker populations on earth (the other is the North Carolina Sandhills population on various ownerships; see 7, Table 5). Of the 25 populations on national forests, at the present time 10 are considered increasing, 10 are stable, and 5 are in decline (Table 4). In contrast, most populations on national forests were declining until the mid 1980’s, and a few were stable (Costa and Escano 1989). Management efforts during the past decade, especially prescribed burning and cavity management, have stabilized most of these populations and led to increases in many. It is

very encouraging that the widespread declines have been stabilized. Our challenge now is to increase the populations to sizes necessary for species recovery.

Current decreasing trends in five populations are the result of several factors (R. Costa, pers. comm.). The critically small population in the Talladega/Shoal Creek Ranger Districts is in decline because of poor habitat conditions resulting from fire suppression and lack of cavity trees, as is the population on the Wakulla Ranger District (James *et al.*, in press). Similarly, the population on the Kisatchie Ranger District is in decline because of inadequate burning, lack of cavities, and overstocked pine stands. The Francis Marion National Forest population is experiencing a decline primarily as a result of complications in the prescribed burning program due to heavy fuel loads from Hurricane Hugo. The decline in the Calcasieu (formerly Vernon) Ranger District is surprising, given the apparent health of the population and its habitat. The reason for this decline is not presently known, but may be the result of differences in field survey and census methods over time, and/or record-keeping. Each of these populations has a substantial role in recovery (below, Table 4; see also 7, Table 5) and these declining trends must be reversed.

Role in Recovery

National forests have a vital role in recovery of red-cockaded woodpeckers, because most core populations within recovery units (see 7) are located in national forests. National forests containing all or part of a primary core population are the Apalachicola (Apalachicola RD), Bienville, Croatan, Francis Marion, Kisatchie (Calcasieu RD), Osceola, Sam Houston, and Talladega (Talladega/Shoal Creek RD). Each of these national forests (or ranger districts) will support a population of at least 350 potential breeding groups at the time and after the species is recovered. National forests containing all or part of a secondary core population are the Apalachicola (Wakulla RD), Catahoula, Conecuh, Davy Crockett, DeSoto (Biloxi and Chickasawhay RDs, separately), Homochitto, Oconee, Ouachita, Sabine, and Talladega (Oakmulgee RD). Each of these national forests (or ranger districts) will support a population of at least 250 potential breeding groups at the time and after the species is recovered. One national forest—the Ocala National Forest in Florida—contains a support population designated essential to recovery of the species because of the importance of conserving red-cockaded woodpeckers in that region. Populations on all other national forests, not designated as primary core, secondary core, or essential support populations, are designated significant support populations (see 7). As federally managed support populations, they are required to be increasing at least until the species is recovered. These populations are valuable because they protect against demographic, environmental, and catastrophic events, contain important genetic resources, and facilitate natural dispersal among populations. Because of these contributions, support populations are necessary to bring the species to recovery but will not be required for species viability once core populations reach population goals identified in delisting criteria (see 6A).

Military Installations

Current Status and Trends

At present there are 16 populations of red-cockaded woodpeckers on military installations (see map insert and Table 4), ranging in size from 2 active clusters (Charleston Naval Weapons Station) to 295 active clusters (Eglin Air Force Base). Eglin's population is currently the fourth largest of all woodpecker populations. Of these 16 populations, 8 are considered increasing, 6 are stable, and 6 have not yet supplied data for trend assessment. Like the populations on national forests, widespread declines among populations on military installations have been stabilized, but substantial increases in population sizes are still required for recovery. However, among various federal agencies, the military is managing red-cockaded woodpeckers most effectively. Rates of increase reported from Marine Corps Base Camp LeJeune and Fort Stewart during the 1990's are among the highest yet documented (in the absence of translocation), an encouraging result of intensive and well-planned management.

Role in Recovery

Military installations have a substantial role in recovery and continuing conservation of red-cockaded woodpeckers. Six military installations contain all or part of six primary core populations: Eglin Air Force Base, Fort Benning, Fort Bragg, Fort Polk, Fort Stewart, and Marine Corps Base Camp LeJeune. These primary core populations will contain at least 350 potential breeding groups at the time of and after the species is delisted. Avon Park Air Force Range is a designated essential support population because it supports one of the largest remaining populations in the ecologically unique South/Central Florida Recovery Unit (see 7). Nine military installations contain support populations, whose increases are important to bringing the species to recovery for reasons described above; however, population goals for these populations are not included in delisting criteria.

National Wildlife Refuges

Current Status and Trends

There are currently 13 populations of red-cockaded woodpeckers partially or wholly contained on national wildlife refuges (see map insert and Table 4), ranging in size from 1 active cluster (Upper Ouachita, Pee Dee, and Black Bayou National Wildlife Refuges) to 118 active clusters (Carolina Sandhills National Wildlife Refuge). Most of these are considered to be stable; two are increasing, two are declining, and several have trends not yet assessed.

Role in Recovery

National wildlife refuges have a small but important role in recovery of red-cockaded woodpeckers. One refuge (Okefenokee National Wildlife Refuge) contains

part of a primary core population, and two refuges contain part of two secondary core populations (Carolina Sandhills and Piedmont National Wildlife Refuges). In addition, two refuges in Northeastern North Carolina (Alligator River and Pocosin Lakes National Wildlife Refuges) contain part of a support population designated essential to recovery because of the importance of conserving red-cockaded woodpeckers in the unique habitat type there. The remaining populations partially or wholly on refuge lands are important or significant support populations (see 7) and should be managed for increasing populations. Big Branch Marsh National Wildlife Refuge, containing 9 active clusters at the present time, is notable among support populations on refuge lands because of its location in an ecoregion (Gulf Coast Prairies and Marshes) that currently contains no other woodpeckers.

Other Federal Lands

Two populations of red-cockaded woodpeckers occur on federal lands other than national forests, military installations, and national wildlife refuges. Big Cypress National Preserve harbors a population of 40 active clusters in the ecologically unique native hydric slash pine habitat of south Florida (see map insert and Table 4). Because of its unique habitat, this population is designated an essential support population. The Savannah River Site, controlled by the Department of Energy, contains an increasing population of 32 active clusters (31 potential breeding pairs) and is a secondary core population (see map insert and Table 4). This population will hold at least 250 potential breeding groups at the time of and after delisting.

In summary, federal lands have a fundamental role in the recovery of red-cockaded woodpeckers. Advances in management of red-cockaded woodpeckers on federal lands have led to stabilization of most populations and increases in many. A few populations are still declining. For most populations designated as primary core, secondary core, or essential support populations, substantial increases are required before recovery population goals are reached.

TABLE 4. Role in recovery, current size, and estimated population trend for red-cockaded woodpecker populations on federal lands, listed by responsible agency. Primary and secondary core populations will contain 350 and 250 potential breeding groups, respectively, at delisting; number of potential breeding groups is equivalent to 0.7 to 0.9 times the number of active clusters. Current size (number of active clusters) is based on 1999 or, in some cases, 1998 breeding season data (USFWS unpublished). Population trend is stable (S), increasing (I), or declining (D) based on definitions given in notes below. NA indicates insufficient data to assess trend.

Agency	Population	Recovery Designation	Current Size (#active clusters)	Current Trend ¹
Forest Service	Angelina NF	Primary Core, part	30	S
	Apalachicola RD	Primary Core	486	S
	Bienville NF	Primary Core	107	S
	Biloxi RD	Secondary Core	6	I
	Calcasieu RD	Primary Core, part	146	D
	Catahoula RD	Secondary Core	31	I
	Chickasawhay RD	Secondary Core	13	I
	Conecuh NF	Secondary Core	14	S
	Croatan NF	Primary Core, part	60	S
	Daniel Boone NF	Significant Support	7	I
	Davy Crockett NF	Secondary Core	51	I
	Evangeline RD	Significant Support	72	S
	Francis Marion	Primary Core	334	D
	Homochitto NF	Secondary Core	45	I
	Kisatchie RD	Significant Support	38 ²	D
	Oakmulgee RD	Secondary Core	123	S
	Ocala NF	Essential Support	18	I
	Oconee NF	Secondary Core, part	17	S
	Osceola NF	Primary Core, part	63	I
	Ouachita NF	Secondary Core	16	I
	Sabine NF	Primary Core, part	25	S
	Sam Houston NF	Primary Core	168	S
	Talladega/Shoal Creek RD's	Primary Core	3	D
	Wakulla RD	Secondary Core	125	D
	Winn RD	Significant Support	16	I
<i>subtotal</i>			2014	10 S, 10 I, 5 D
Fish and Wildlife Service	Alligator River NWR	Essential Support, part	2	NA
	Black Bayou NWR	Important Support	1	NA
	Big Branch Marsh NWR	Significant Support	9	NA
	Carolina Sandhills NWR	Secondary Core, part	118	I
	D'Arbonne NWR	Important Support	4	D
	Felsenthal NWR	Significant Support	15	D
	Noxubee NWR	Significant Support	38	S
	Okefenokee NWR	Primary Core, part	29	S
	Pee Dee NWR	Important Support	1	S
	Piedmont NWR	Secondary Core, part	37	I
	Pocosin Lakes NWR	Essential Support, part	4	S
	St. Marks NWR	Essential Support	6	S
	Upper Ouachita NWR	Important Support	1	NA
<i>subtotal</i>			264	5 S, 2 I, 2 D,
<i>Table continued next page.</i>				4NA

TABLE 4 (cont.). Role in recovery, current size, and estimated population trend for red-cockaded woodpecker populations on federal lands

Agency	Population	Recovery Designation	Current Size (#active clusters)	Current Trend ¹
Military:				
Air Force	Avon Park AFR	Essential Support	21	S
	Dare Co. Bombing Range	Essential Support, part	9	NA
	Eglin AFB	Primary Core	295	I
	Poinsett Weapons Range	Significant Support	6	NA
Army	Camp Mackall	Primary Core, part	11	NA
	Fort Benning	Primary Core	186	I
	Fort Bragg	Primary Core, part	350	I
	Fort Gordon	Significant Support	3	I
	Fort Jackson	Significant Support	21	I
	Fort Polk	Primary Core, part	44	I
	Fort Stewart	Primary Core	198	I
	MOT Sunny Point	Significant Support	6	NA
	Peason Ridge	Significant Support	27	NA
Marine Corps	MCB Camp LeJeune	Primary Core, part	50	S
Navy	Charleston Naval Weapons Station	Significant Support	2	NA
<i>subtotal</i>			1218	2 S, 7 I, 6 NA
Dept. of Energy	Savannah River Site	Primary Core	31	I
<i>subtotal</i>			31	1 I
National Park Service	Big Cypress NP	Essential Support	40	NA
<i>subtotal</i>			40	1 NA
Total			3579	17 S, 20 I, 7 D, 11 NA

¹Population trend is increasing if average annual percent change in number of active clusters, 1995-1999, is ≥ 5 percent. Population trend is declining if (1) there was a 10 percent drop in the number of active clusters from one year to the next (within the last 3 years), or (2) if the number of active clusters decreased for 3 consecutive years (within the last 5 years). Population trend is stable if neither increasing nor declining.

²Maximum number of active clusters from a district-wide comprehensive survey in May 2000 (USFWS unpublished).